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Influence of controlled burning on the mobility and temporal variations of potentially toxic metals (PTMs) in the soils of a legacy gold mine site in Central Victoria, Australia

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ABSTRACT

Controlled burns also known as managed burns or prescribed fires conducted in fire-prone areas are an efficient and economic option to reduce the frequency and intensity of wildfires. The objective of this study is to investigate the remobilization of potentially toxic metals (PTMs) in the soils of a legacy gold mine site in Central Victoria, Australia after a controlled burn and to describe their temporal variations in concentrations. Soil samples were collected two days before, two days after and five times later (3, 6, 9, 12 months and after major rainfall) in the post-burn environment after a controlled burn, from the Maldon legacy mine site and analysed for PTM concentrations. The results revealed PTM mobilization after the burn and most of the PTMs (As, Cd, Cu, Mn, Ni, and Zn) evidenced an increase immediately after the burn but a reduction in the subsequent post-burn environment. The increase is postulated to be associated with addition of PTM enriched ash to the soil, while the decrease is due to the removal of ash and surface soil by wind activity as well as rainfall runoff and leaching. The PTM mobility is of specific concern due to the negative impacts on human and ecosystems health. Climate change and the resulting projection of increased forest fire frequency highlight the environmental significance, given the expected concomitant increase in PTM mobilization through wildfires and controlled burns. Hence, the practice of controlled burning should be carefully considered as a forest management option in any legacy mining areas and indeed in other areas where PTM contamination is reported.

1. Introduction

Potentially toxic metal (PTM) (As, Cd, Co, Cr, Cu, Mn, Ni, Pb and Zn) contamination associated with mining and mineral processing, places a significant financial and environmental burden on communities around the world (Pearce et al., 2010; Esshaimi et al., 2012; Cobbina et al., 2013; Martin et al., 2013; Schaider et al., 2014; Martin et al., 2016, 2017). This is predominantly because of the negative impacts of PTMs on human and environmental health, particularly their toxicity, bioaccumulation and biomagnification tendency in the food chain and the environmental persistency (Adriano, 2001; Ignatavièius et al., 2006). PTM contamination is a significant problem at legacy or historical mining areas as well, due to the lack of environmental regulations during mining and following closure resulting in the accumulation of millions of tonnes of mine waste materials rich in PTMs. The volume of the land affected by PTMs makes this a significant global issue as more than a million legacy mine sites estimated to exist across the globe (UNEP, 2001; Lottermoser, 2007; Erickson et al., 2008; Zhou et al., 2010; Esshaimi et al., 2012; Zornoza et al., 2012; García-Lorenzo et al., 2014; Armienta et al., 2016).

Soil PTM contamination from both existing and legacy mine sites has been reported from around the world. Harrison et al. (2003) investigated the PTM contamination around the Yerranderie legacy Ag-Pb-Zn mine site in New South Wales, Australia and reported higher PTM (Ag, As, Cd, Cu, Hg, Pb, and Zn) concentrations in soil surrounding the mine site, which are up to 400 times the guideline values. Soil core sample analysis showed that an overall increase occurred in and around 1950, which is coincident with high rainfall and preceded drought periods (Harrison et al., 2003). Similarly, Schaider et al. (2007) studied the levels of Zn, Pb and Cd in a legacy mine site (Tar Creek Superfund site) in Oklahoma, USA and found elevated concentrations of Zn ($9100 \pm 2500\ mg\ kg^{-1}$), Pb ($650 \pm 360\ mg\ kg^{-1}$) and Cd ($42 \pm 10\ mg\ kg^{-1}$) in soil samples, which are relatively labile and are present in bio-accessible mineral phases. In a similar manner, Esshaimi et al. (2012) investigated soil and water contamination in the vicinity of the Kettara abandoned mine, south Morocco and reported high concentrations of Cu, Pb and Zn in the tailings (76, 80, $79\ mg\ kg^{-1}$ respectively) and in soil (68, 52 and $26\ mg\ kg^{-1}$ respectively) samples.

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Moreover, PTM dispersion to downstream and downslope areas was also observed, and this was considered to be due to the surface runoff and strong wind actions (Esshaimi et al., 2012). Other studies such as Clark et al. (2001) from Australia, Zornoza et al. (2012) from Spain, Lee et al. (2001, 2005) from South Korea, Zhou et al. (2010) from China and Cobbina et al. (2013) from Ghana are all significant in this context, as each of these study sites documents the extent of contamination of soil, sediments, water bodies and general vegetation with PTMs from legacy mine sites.

Central Victoria in Australia, is not exempt from PTM contamination as it has seen extensive mining since 1850, and has produced > 2500 t of gold with resulting accumulation of tonnes of mine waste materials rich in PTMs (Sultan, 2006, 2007; Pearce et al., 2010, 2012; Martin et al., 2013, 2014, 2017). Any disturbance of this contaminant pool is of concern, since it will act as a catalyst for enhanced mobility and dispersion of PTMs. One such major agent of disturbance is forest fire, which includes controlled burns (Certini, 2005; Neary et al., 2005; Pereira et al., 2011; Abraham et al., 2017a, 2017b; Alcañiz et al., 2018).

Fires are natural process in many terrestrial ecosystems and some are severely destructive in nature to the vegetation and surface soil (Neary et al., 2005; Beganyi and Batzer, 2011). The human-fire relationship goes back to the pre-historic period, where humans used fire in preparing food, hunting and agricultural land preparation (Pyne, 2016). Recent studies show the adverse and diverse impacts of fire to the environment, economic sustainability and social structures (Smith et al., 2011; Liu et al., 2014). For example, fire may disturb the forest ecosystem structure and functions by affecting the nutrient cycling, vegetation distribution and composition and herbivore distribution (Augustine et al., 2014; Alcañiz et al., 2018), however, some ecosystems are fire dependent (Alcañiz et al., 2018). Fire may alter a range of substantial physical and biogeochemical soil properties and affect the catchment behaviour by affecting key hydrological factors such as infiltration and runoff (Certini, 2005; Shakesby and Doerr, 2006; Knoepp et al., 2008; Verma and Jayakumar, 2012; Abraham et al., 2017b). The contaminant mobilization associated with forest fire is of global significance but region specific (Jovanovic et al., 2011; Costa et al., 2014; Campos et al., 2015, 2016; Abraham et al., 2017a, 2017b). The fire potentiality depends on the increased fuel load, dry weather patterns with increased atmospheric temperature and wind speed, all of which enhance ignition potential (Certini, 2005; Hennessy et al., 2005; Westerling et al., 2006; Alcañiz et al., 2018). Recent climate change studies highlight the increase in frequency and areal extend of fire in many parts of the world in association with rural depopulation, land abandonment and afforestation with fire-prone species (Hennessy et al., 2005; Westerling et al., 2006; Moreira et al., 2009; IPCC, 2013; Campos et al., 2015, 2016). This resulting fuel-rich environment consequently requires added controlled burning events as a fire risk mitigation strategy.

Controlled' or 'prescribed burning' is a type of managed forest fire, where deliberate application of fire, acts primarily to reduce the available fuel load and to reduce the frequency and intensity of wildfires (Hatten et al., 2005; Pereira et al., 2011; Fernandes et al., 2013; Bennett et al., 2014; Alcañiz et al., 2018). This low-to-moderate intensity fire event reduces the fuel load by consuming the understory vegetation and part of the forest floor layers, and has long been considered as an efficient and economic option for fire control (Fernandez and Botelho, 2003; Pereira et al., 2011; Alcañiz et al., 2018). Controlled burning is also used for agricultural land preparation, weed and insect control, biodiversity management among other land management applications (Pereira et al., 2011; Fernandes et al., 2013) and is widely practised in fire-prone areas in the American, European and Australian continents (Wade et al., 1989; Fernandez and Botelho, 2003; Certini, 2005; Castellinou et al., 2010; Pereira et al., 2011). Although it has a number of benefits, controlled burn constitutes a disturbance on the forest environment, alters the ecological processes and affects the properties of the surface soil (Santín and Doerr, 2016; Alcañiz et al., 2018). Moreover, precipitation and subsequent runoff events in the post-burn environment may result in the transport of surface soil and ash, which contains elevated levels of various contaminants, including PTMs to downslope and downstream surface water resources (Ignatavièius et al., 2006; Burke et al., 2010; Stein et al., 2012; Burke et al., 2013; Costa et al., 2014; Abraham et al., 2017b). This has heightened concern that this series of events may act as a contaminant source for potable water resources serving local communities and may negatively impact the health of ecosystems (Ignatavièius et al., 2006; Burke et al., 2010; Smith et al., 2011; Stein et al., 2012; Burton et al., 2016; Abraham et al., 2017a, 2017b).

Information regarding PTM mobility in the environment is vital for ecosystem management and sustainability due to the potential environmental persistence, toxicity and health impacts on human and ecosystems, together with their potential bioaccumulation characteristics (Adriano, 2001; Ignatavièius et al., 2006; Abraham et al., 2017a, 2017b). Wildfires have been found to mobilize PTMs in the atmospheric, terrestrial and aquatic compartments mostly by the combustion of vegetation and the subsequent release of ash and the burning of soil organic matter (Costa et al., 2014; Kristensen et al., 2014; Campos et al., 2015, 2016; Abraham et al., 2017a, 2017b, 2018). While some studies report the influence of wildfire on PTM mobilization (Odigie and Flegal, 2011, 2014; Kristensen et al., 2014; Burton et al., 2016; Campos et al., 2015, 2016; Odigie et al., 2016; Nunes et al., 2017; Wu and Taylor, 2017), no studies have been reported from a legacy gold mine site, associated either with wildfire or controlled burn. Since, wildfire has the potent to release and remobilize PTMs, we hypothesis that a controlled burn is also able to release and remobilize PTMs, specifically from a legacy mining landscape, even though the burning temperature is comparably low. Hence, the current study is designed (i) to identify the mobility and distribution of PTMs in soil, immediately after controlled burn event and (ii) to identify the temporal variations of PTMs concentrations in the post-burn soil environment. This study provides geochemical information to land and water managers to inform their decision-making to achieve better environmental and health outcomes, including water quality in fire-prone areas (specifically mining affected and PTM affected landscapes) in Australia and in number of countries across the globe.

2. Materials and methods

2.1. Study area

Central Victoria in Australia has hundreds of legacy gold mine sites and most of them are located in forest areas. The study area Maldon is a legacy gold mining township, located 145 km NW of the City of Melbourne and 40 km SW of Greater Bendigo in Victoria, Australia. In the Maldon area, gold mining started in 1853 and the phase of deep mining ceased in 1926, mainly because of difficulties in the mining process, but later (in 1990s) mining restarted in the area with modern technology (Parkweb, 2018). Cradle and gold pan to separate gold from alluvium was the early gold extraction techniques, but later moved to puddling, quartz reef mining, deep lead mining and various forms of sluicing, among them quarts reef mining was the widespread technique (Egold, 2018). Open cut trenches, shafts and adits, which used to access sulphide bearing ores can be seen in the area as mining remnants (Cherry and Wilkinson, 1994; Maldon, 2017). During this early mining period, the area produced 56 t of gold from various mining sites (Cherry and Wilkinson, 1994). Mining and the lack of environmental regulations during historical mining period, resulted in the accumulation of tonnes of mine waste materials in the area, which are rich in PTMs (Sultan, 2006, 2007; Martin et al., 2013, 2014). From an area with a number of legacy mine sites, one, which was located on public land, east of the Union Hill mine was selected for controlled burn related PTM mobility study (Fig. 1). While the study area is a public forest land (State forest) under the jurisdiction, it is located near the Maldon town

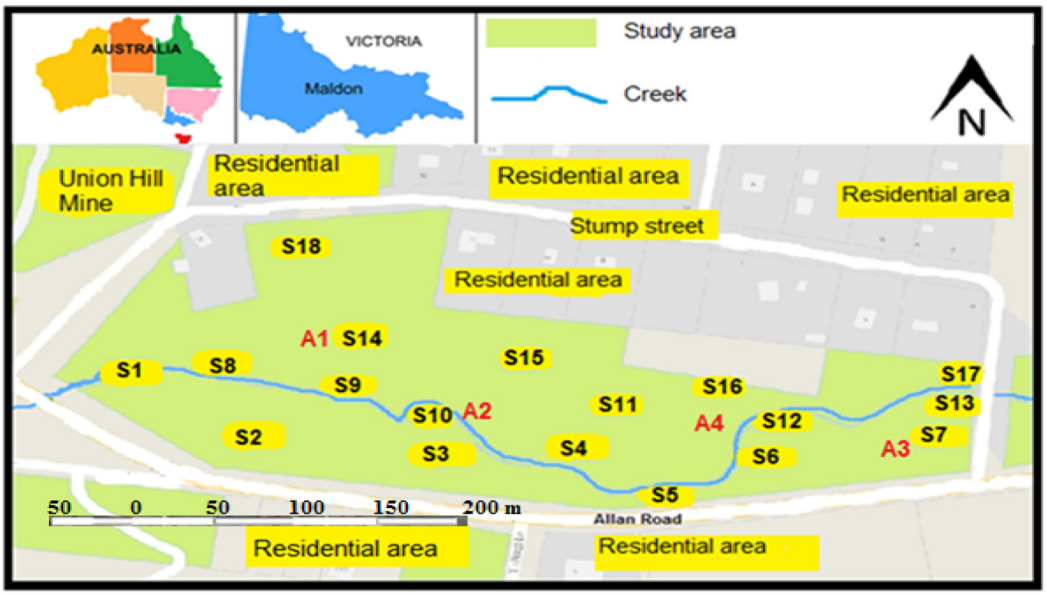

Fig. 1. Location of the study area with soil sampling (S1 to S18) and ash sampling (A1 to A4) stations identified.

centre, and is thus close to a residential area.

The area receives an average annual rainfall of 540 mm, with cold wet winters (7.5–20.5 °C) and warm dry summers (16.4–40.3 °C) (Taylor and Alley, 2001; BoM, 2018). A creek running from west to east channels much of the rainfall falling in the study area, which is significant as it may carry all contaminants including PTMs to the downstream water resources. The canopy in the area is native acacia (*Acacia pycnantha*) (*60%*), and eucalyptus (*Eucalyptus radiata*) (*40%*), whereas, the understory species are the introduced gorse (*Ulex europaeus*) (70%), Chinese shrub (*Cassinia arcuata*) (10%), golden shrimp (*Pachystachys lutea*) (10%) and Onion weed (*Asphodelus fistulosus*) (10%).

A controlled burn was conducted in the area in late August 2015 (end of winter) for fire risk mitigation and land management (weed control) purposes by the Department of Environment, Land, Water and Planning (DELWP) of the State of Victoria and Parks Victoria fire crews, and we capitalised this opportunity to study PTM mobility. The Parks Victoria fire crews ignited the fire around 10:00 am at the western part of the study area and moved to the east. Immediately after the ignition, fire grew quickly because of the presence of high dry fuel load and the smoke moved to the SW in conjunction with wind direction. The fire observed as 'seemingly intense' for several minutes due to the observed rise in flame, around 2 m in height (Fig. 2). It took around one and half hours to burn the major fuel loads in the area, followed by slow burning of plant materials which continued for two days. The fire burned most of the ground cover litter and shrub type vegetation (< 2 m in height). Around 80% of the study area was affected by the burn and the government agency responsible for the burn had no history of previous fire use in this area (DELWP pers comm). There were large amounts of ash observed close to some of the experimental sampling stations (S13, S14 and S10) where high fuel load and tree trunks (down wood) were burned. In some spots, it was grey or light grey in colour, while other spots mostly black in colour. The fire consumed the shrubs completely, but had limited impact on the canopy.

There was no rainfall in between before burn (2DBB- two days before burn) (first) and immediately after burn (2DAB – two days after burn) (second) sampling occasions, however the cumulative rainfall in between the second and third (3MAB- three months after burn) sampling period was 79 mm and between third and fourth (6MAB- six months after burn) sampling period was 100.2 mm respectively. Between the fourth and fifth (9MAB- nine months after burn) sampling period, the cumulative rainfall was 164 mm and between fifth and sixth (12MAB- twelve months after burn) sampling period received the highest rainfall of 243 mm. However, the rainfall after the sixth sampling period is defined as the major event with 155 mm rainfall within a fortnight. The total monthly rainfall and mean monthly temperature are given in Fig. 3 (BoM, 2018).

2.2. Soil sampling

The nature and composition of surface soil is reported to be considerably affected by the intensity and duration of fire. In addition, during the post-fire environment, the soil is susceptible to human intrusion, wind-borne transport of fine materials and to rainfall and surface runoff (Hernandez et al., 2003; Mandal and Sengupta, 2006; Schaider et al., 2007; Shcherbov, 2012). With these exogenous influences in mind, in order to assess the extent of PTM mobility in the surface soils after a controlled burn and to examine their dispersal in the post-burn environment, 18 surface soil samples (variable depth up to 3 cm) were collected at one time. These samples were gathered (i). two days before burn (2DBB), (ii). two days after burn (2DAB), (iii). 03/06/09/12 months after burn (3MAB/6MAB/9MAB/12MAB) and (iv). immediately after a major rainfall event (IAMR) (155 mm within a fortnight) in September 2016 (two weeks after the 12MAB sampling). All the seven sets of samples were collected from the same sampling stations (S1 to S18, noted in Fig. 1), using the STAR sampling method with an established trace metal clean technique procedures (Ritson et al., 1999; Soto-Jimenez et al., 2006; Taylor et al., 2010; Odigie and Flegal, 2011). STAR sampling is a soil sampling method in which surface soil samples were collect at the centre and five corners of a STAR (in shape) in a location and mixed together before transfer to the sampling container. Approximately 1 kg of bulk surface soil samples (up to 3 cm of depth) were collected within an area of 1600 cm^2 (40 cm × 40 cm) from each sampling station (S1 to S18) with STAR method and were transferred to clean and dry polyethylene containers at the sampling site and brought to the Federation University Australia laboratory for analysis. A total of 126 soil samples (18 stations × 7 sampling times) were collected over a period of one year from the study area.

Fig. 2. Showing the fire intensity during the controlled burn.

2.3. Ash sampling

Ash is defined as the organic and inorganic residue formed by the burning action of fire and the colour of the ash can provide some indication of the fire intensity (Bento-Gonçalves et al., 2012). White coloured ash denotes complete combustion of the fuel load in a high intense fire environment, whereas, grey ash represents a moderate to high fire intensity (Úbeda et al., 2009). In the study area, after fire, patches of grey and light grey ash were observed, but most of the ash was black in colour consistent with a low to moderate intensity fire (Úbeda et al., 2009; Bento-Gonçalves et al., 2012). The thickness and colour of ash covering the soil surface after the fire varied in space and time. This heterogeneous distribution of ash is mainly due to the types of fuel load, amount of fuel, burning temperature, duration of burning, wind pattern and different degree of combustion completeness (Bodí et al., 2014). During the controlled burn, thick ash layers were formed where thick fuel loads were available. A total of four ash samples (each of approximately 200 g) were opportunistically collected immediately after the burn, from locations (Fig. 1) were thick ash (> 1 cm) were available, using the trace metal clean technique procedures. Since ash samples were collected only from the thick ash locations so as to limit any soil or other contamination. The ash samples were stored in glass containers and transported to the Federation University Australia soil laboratory for analysis.

2.4. Sample analysis

In the soil laboratory, soil and ash samples were transferred to aluminium trays and placed in the oven for 24 h (below 60 °C). The oven dried samples were sieved through the 2 mm wire mesh sieve, and < 2 mm fractions were used for analyses. Soil and ash pH and electrical conductivity (EC) were determined with US-EPA 9045D and 9050A analytical procedures respectively, and organic matter (SOM/AOM) and carbonate content (both expressed as percentage) were determined with the Loss on Ignition (LOI) method as detailed mentioned in Rayment and Lyons (2014).

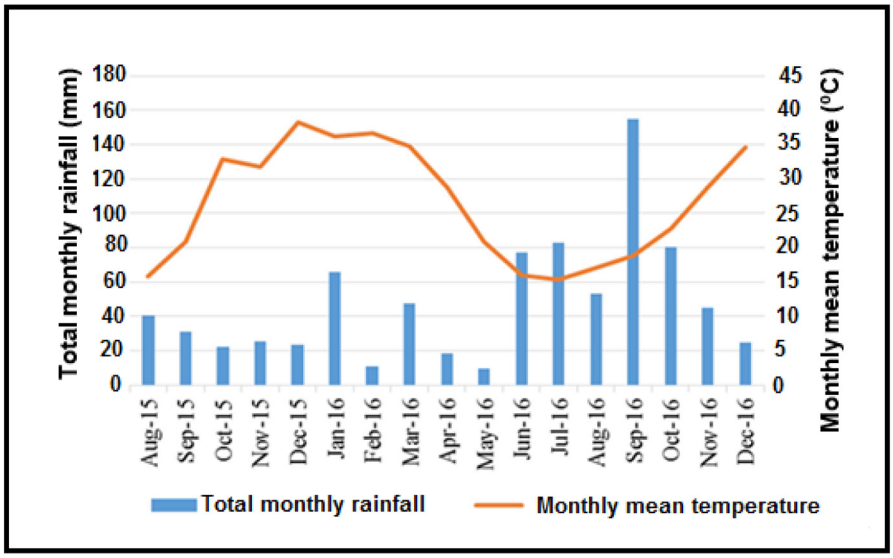

Fig. 3. Total monthly rainfall (mm) and mean monthly temperature (°C) at the Maldon study area during the first 16 months after the controlled burn in late August 2015 (BoM, 2018).

All PTMs analysis in the soil samples were conducted in a NATA (National Association of Testing Authorities) accredited laboratory (Accreditation No. 825) by Ultra-Trace Level Methods using ICP-MS (Inductively Coupled Plasma-Mass Spectrometry) and ICP-AES (Inductively Coupled Plasma-Atomic Emission Spectrometry) with Aqua Regia Digestion (GEO-AR01). Oven dried and sieved (using stainless steel wire mesh) soil samples were milled using stainless steel bowls, and 0.5 g of the milled sample was used for PTM analysis. Aqua regia was used to digest the sample on a graphite heating block and allowed to cool. The cooled solution was mixed with deionised water and analysed by ICP-AES. Results that returned a higher than scoped concentration of Hg were diluted accordingly. These diluted samples were then analysed by ICP-MS for the remaining set of metals and the results were corrected for inter-elemental spectral interference (Santoro et al., 2017).

2.5. Data analysis

The data were statistically analysed using IBM SPSS (version 23) and R (version 3.4.1) software(s). Descriptive statistics including minimum, maximum, median, percentiles and interquartile range (IQR) were obtained for the following soil properties: pH, EC, SOM, carbonate content and PTM concentrations in the soil in the pre- and post-burn environments. Wilcoxon signed rank sum tests were conducted to determine if statistically significant differences existed between (i) the median concentrations of each PTM and (ii) the physicochemical characteristics of the soil immediately before and after the controlled burn, and immediately before and after the rainfall event. Friedman's test has applied to test the statistical significance of the influence of the time (03/06/09/12 months) on the soil pH, EC, SOM, carbonate content and PTM concentrations in the soil. The Spearman rank correlation coefficient was used to relate the concentration of physicochemical properties of the soil to the PTMs in the soil, both from the corresponding sampling periods and the metals themselves.

3. Results and discussion

Pre- and immediate post-burn soil data from the Maldon legacy mine site (study area) were collated and compared in terms of pH, EC, SOM, carbonate content and PTM concentrations, in order to understand the effect of controlled burn on the physicochemical characteristics of the soil and the PTM concentrations in the soil. All data from the post-burn environment were also analysed to determine the influence of time, specifically the effect of major rainfall (in September 2016 after a year of controlled burn) on the soil physicochemical characteristics and the PTM concentrations in the soil.

3.1. Immediate effects of controlled burn on soil characteristics

A summary of the physicochemical characteristics (pH, EC, SOM and carbonate content) of soil, two days before and two days after the controlled burn, followed by five times (four times of sampling at three months intervals and the last sampling was two weeks after the previous and immediately after a major rainfall event). A combined data set spanning more than a year is measured and the relevant details are presented in Fig. 4 and in a Supplementary table (Table S1). The immediate post-burn soil samples demonstrated significantly higher median values of pH ($p < 0.05$) and EC ($p < 0.01$), and higher SOM ($p < 0.08$) compared with pre-burn soil samples with differences in these values amounting to a factor(s) of 1.1, 2.3, and 1.7 respectively.

Immediately after the controlled burn, the median pH of the soil (4.8) was significantly higher ($p < 0.05$) than that of the pre-burn soils (4.3) with values differing up to 1.9 times were observed. The minimum observed pH in the pre-burn environment was 3.6 and the maximum was 5.6, whereas in the post-burn environment, the observed minimum was 3.8 and the maximum increased to 8.6. Among the 18 sample stations in the study area, 12 stations (66%) showed an increase in pH value immediately after the burn.

Similarly, immediately after the burn, median EC of the soil (863 $\mu S\,cm^{-1}$) was significantly higher ($p < 0.01$) than that of the pre-burn soil (375 $\mu S\,cm^{-1}$), with values differing up to 5.6 times greater were observed. The minimum EC observed in the pre-burn environment was 95 $\mu S\,cm^{-1}$ and the maximum was 970 $\mu S\,cm^{-1}$, but in the immediate post-burn environment, the minimum observed increased to 174 $\mu S\,cm^{-1}$ and the maximum increased to 2800 $\mu S\,cm^{-1}$. Among the 18 sampling stations, 13 stations (72%) showed increases in EC values, and five stations (28%) showed decrease in values immediately after the burn. The highest EC variation observed was 2011 $\mu S\,cm^{-1}$, followed by 2005 $\mu S\,cm^{-1}$, and the lowest being 28 $\mu S\,cm^{-1}$.

The SOM (median) almost doubled (15.2%) immediately after burn, compared to the pre-burn median value (8.7%). Similar to pH, 12 sampling stations (66%) demonstrated increase in SOM content and the remaining (34%) stations shown decrease in concentration. The minimum SOM value in the pre-burn environment was 4% and the maximum was 42.3%. Even though, median SOM highlighted increase, the minimum value reduced to 2.7%, and the maximum reduced to 33.3% in the immediate post-burn environment. The highest increase observed was at S3 and the lowest was S17, whereas, the highest decrease observed was at S4 and the lowest decrease was at S18.

The median carbonate value in the immediate post-burn environment did not show any significant variation in concentration compared with the pre-burn median value. However, if we take individual sample stations, S4 station records the highest increase in concentration and S13 shown lowest increase. Among the 18 stations, 10 stations shown reduction on carbonate value and the highest reduction observed was at S11 and lowest deduction observed was at S18.

The fire-induced increases in soil pH and EC, noted immediately after the burn are consistent with existing studies (Ulery et al., 1993; Hernández et al., 1997; Úbeda et al., 2005; Quintana et al., 2007; Granged et al., 2011; Campos et al., 2015, 2016). The increase is considered to be due to the combustion of vegetation with subsequent deposition of ash, which adds large amounts of alkaline oxides and hydroxides to the surface soil (Arocena and Opio, 2003; Certini, 2005; Verma and Jayakumar, 2012; Alcañiz et al., 2018). The decrease in pH also relates to the reduction in the concentration of organic acids (Ulery et al., 1993, 1995; Verma and Jayakumar, 2012; Campos et al., 2016). The high pH and EC of ash samples compared to the surrounding soil samples were consistent with this postulate and similar results were reported earlier (Naidu and Srivasuki, 1994; Kim et al., 2003; Goforth et al., 2005; Campos et al., 2016). In the study area, the major fuel load was understory vegetation gorse, Chinese shrub, golden shrimp, onion weed and the leaf litters of canopies such as eucalyptus and acacia and were burned almost completely, resulting in the deposition of thick ash layer in specific places where fuels were available in high quantity. Other sampling locations, ash dispersed by burning and wind activities. The sampling stations, which shown high pH variations were close to the thick ash production sites.

Several laboratory studies address the behaviour of soil pH during fire (Sever et al., 2001; Badía and Marti, 2003; Certini, 2005; Marcos et al., 2007; Moreno et al., 2015; Thomaz, 2017). However, the laboratory studies largely ignore the production of ash and the interaction of ash with soil as most of the laboratory studies are conducted by heating the soil in controlled environment. Moreover, the laboratory studies use residence time much longer than those experienced by soil during the forest fire. Fluctuation in temperature which is highly likely in a real fire environment is also difficult to model in laboratory studies. Even though, laboratory studies can't model the real time fire environment, most of the studies highlighted the increase in soil pH according to the rise in temperature (Badía and Marti, 2003; Certini, 2005; Marcos et al., 2007; Moreno et al., 2015; Thomaz, 2017).

Low severity controlled burns result in only a slight change in SOM concentration, while high severity wildfires results in a more significant

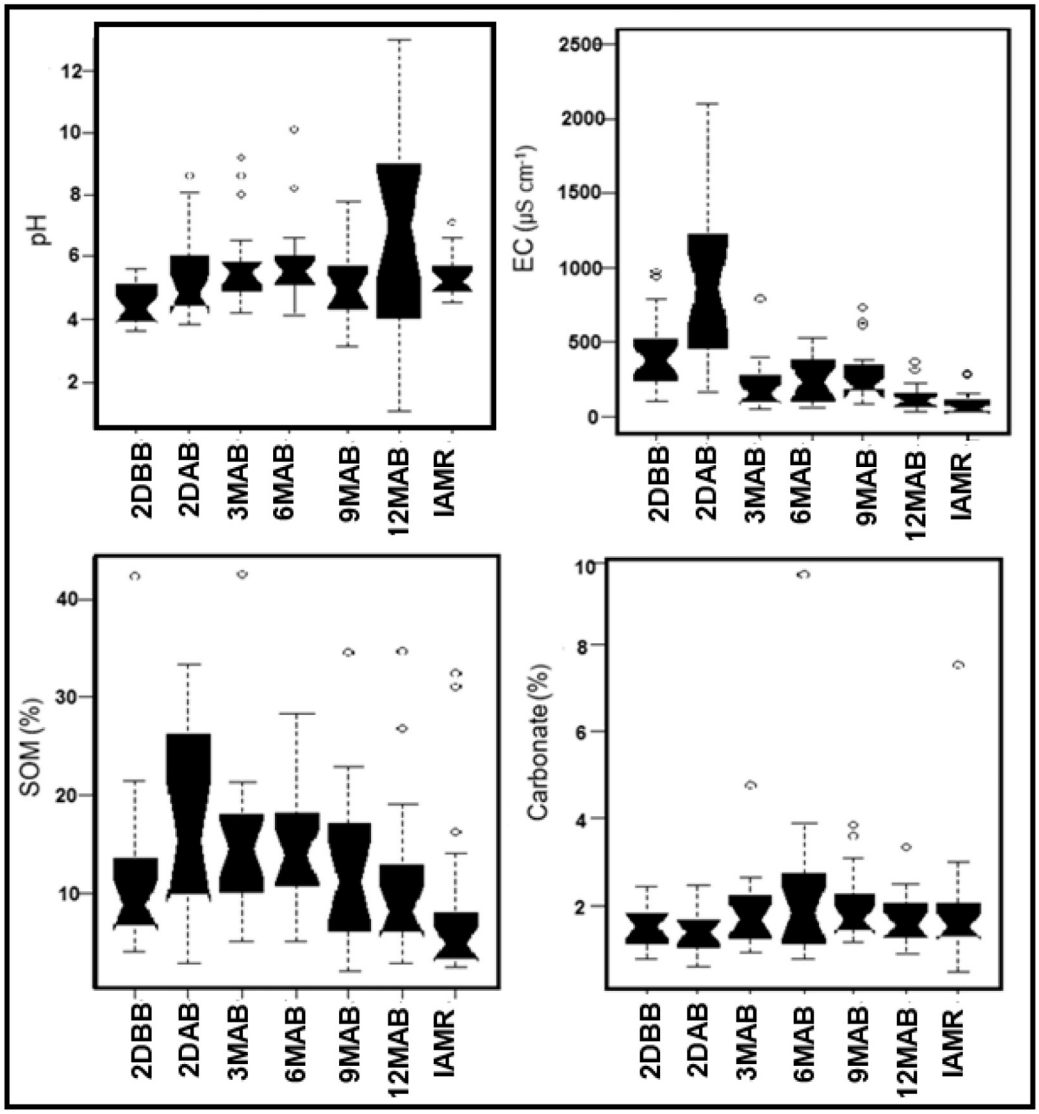

Fig. 4. Notched boxplot showing the concentrations of pH, EC (µS cm^{-1}), SOM (%) and carbonate content (%) in various sampling occasions (2DBB- two days before burn, 2DAB- two days after burn, 3MAB -three months after burn, 6MAB-six months after burn, 9MAB-nine months after burn, 12MAB-twelve months after burn, and IAMR- immediately after major rainfall in September 2016) (n = 18). Note the increase in pH, EC and SOM immediately after the fire.

alterations (Johnson, 1992; Bogacz et al., 2011; Verma and Jayakumar, 2012). The effect of fire on SOM is highly variable, and depends on (i) the type and intensity of the fire, (ii) the nature of fuels, (iii) prevailing soil moisture levels, and (iv) other physical attributes, so that generalizations are challenging (González-Pérez et al., 2004; Verma and Jayakumar, 2012). In this study, immediately after the controlled burn, median SOM (15.2%) were higher than that of the pre-burn soils (8.7%), where it was observed that values differed up to 3.8 times across the samples taken. This increase in SOM is in agreement with existing studies (Pardini et al., 2004; Jovanovic et al., 2011; Faria et al., 2015; Campos et al., 2016) and is considered to be due to the accumulation of pyrogenic carbon (partially charred leaves, litter, forest necromass and the deposition of ash) formed by the incomplete combustion of biomass (González-Pérez et al., 2004; Goforth et al., 2005; Jovanovic et al., 2011; Bodí et al., 2014; Faria et al., 2015).

After the burn, 8 stations showed a slight increase (1.3%) and 10 stations showed a small decrease (1.5%) in carbonate content. The limited variation is probably due to the capability of carbonate to resist temperatures of up to 1000 °C (Rabenhorst, 1988). When the burning temperature exceed 580 °C, the carbonate may be converted into oxides with distinct morphological changes from massive to sponge-like being reported (Kloprogge et al., 2004). New carbonate can be formed by the interaction of ash with atmospheric CO_2 and moisture resulting in the increase in particle size due to agglomeration (Steenary et al., 1999) or swelling of ash particles (Stoof et al., 2010; Bodí et al., 2014). However, in this study, the lack of substantive change in carbonate concentration is attributed to the low burning temperature (below 580 °C).

3.2. Soil physicochemical characteristics – temporal variations

It is reported that both soil pH and EC increase after a forest fire, but they return to the pre-burn level after several months as observed in many previous studies (Ulery et al., 1995; Hernández et al., 1997; Kim et al., 2003; Granged et al., 2011). In this study, three months after burn (3 M) samples displayed a minor increase (compared to the immediately after burn samples (IAB)) in median pH from 4.8 to 5.5 with a maximum of 3.7 units. The pH is similar in six months after burn (6 M) samples (5.5) with a maximum observed increase of 1 unit only. However, in nine months after burn samples (9 M), it showed a reduction from 5.5 to 4.9 (with the observed maximum of 2.3 units), followed by an increase in twelve months after samples (12 M) (5.7). The increase in 12 M after burn is considered to be due to the increase

in rainfall in winter and the subsequent interaction of ash with the surface soil. Compared to 12 M samples, the samples collected immediately after the major rainfall (IAMR) highlighted a slight reduction in pH (5.2). All variations noted here in pH levels were statistically significant ($p < 0.01$).

The controlled burn doubled the median EC value of the surface soil (from 375 to 863 μS cm^{-1}), which displayed a reduction after 3 M sapling (155 μS cm^{-1}) (with a maximum observed decrease of 2047 μS cm^{-1}). After the 6 M samples, the median EC increased from 155 to 228 μS cm^{-1} (with a maximum observed increase of 1160 μS cm^{-1}), but in 9 M, the mean EC reduced to 196 (with a maximum observed decrease of 3530 μS cm^{-1}). A further median EC reduction was observed in 12 M samples (102 μS cm^{-1}) and after the major rainfall event (50 μS cm^{-1}). The mean EC value dropped below the pre-burn value in 3 M sampling onwards until after the rainfall sampling, displaying the ephemeral characteristic of EC. All EC reductions are considered to be caused by the rainfall events and all variations are noted as statistically significant ($p < 0.001$).

In the post-burn environment SOM demonstrated statistically significant ($p < 0.01$) reduction (IAB-15.2, 3 M-14.4, 6 M-13.8, 9 M-11.1, 12 M-8.1 and IAR-4.8%) in all the sampling periods and in 9 M samples it fell below the pre-burn level of 8.7%. Compared to pH, EC and SOM, variation of carbonate is negligible in the post-burn environment (variations are from 1.4, to 1.7, 1.8, 1.7, 1.6 and 1.5% only).

The post-burn temporal reductions in pH, EC and SOM are in agreement with other studies (Hernández et al., 1997; Kim et al., 2003; Granged et al., 2011; Campos et al., 2016), and were attributed to the influence of wind erosion as well as rainfall and subsequent runoff, which may have transported ash and surface soil downslope, downstream and down wind direction (Ulery et al., 1993; Zabala et al., 2014; Campos et al., 2016). The increase in soil pH during 3 M and 6 M after burn is probably due to the continued mixing of ions from the existing ash with surface soil, and the reduction in 9 M after burn is probably associated with the leaching effect of the increase in rainfall (163 mm) similar to that reported by Costa et al. (2014). The decreases in EC during 3 M and 9 M are also explained by the effects of rainfall, while the increase in levels in 6 M compared to 3 M and 9 M may be due to the lack of rainfall in summer.

3.3. Physicochemical characteristics of ash

Bodí et al. (2014) defined "Forest fire ash is the particular residue, remaining on the ground from the burning of forest fuels consisting of mineral materials and charred organic components". It is a complex mixture of organic and inorganic residues, include soil materials present in the litter and vegetation before fire. Ash may also contain burned soils, which has lost its structure, oxides and hydroxides of base cations, silica, phosphorous, nutrients, major and trace metals and potential contaminants produced by fire. (Goforth et al., 2005; Plumlee et al., 2007; Gabet and Bookter, 2011; Bento-Gonçalves et al., 2012; Bodí et al., 2014; Campos et al., 2016; Abraham et al., 2017a, 2017b). These materials may be subsequently deposited on the soil surface in burned areas. Ash can provide valuable information about the fire severity and consumption completeness of fuels (Keeley, 2009; Pereira et al., 2014). The low to moderate severity burns result in black ash, whereas, high fire severity results in white or grey ash (Pereira et al., 2014). The presence of various ions in ash generally correlates with alkaline nature and elevated EC (Ulery et al., 1993; Gabet and Bookter, 2011). The pH, EC, OM, carbonate content and the colour of four ash samples collected just after the controlled burn, are summarised in Table 1.

The ash samples demonstrated relatively high pH (ranges from 9.1 to 11.6) and EC (ranges from 1173 to 28,500 μS cm^{-1}) similar to previous studies (Kim et al., 2003; Goforth et al., 2005; Úbeda et al., 2009; Pereira et al., 2011, 2013; Campos et al., 2015, 2016). The high pH and EC values of ash (compared with soils) immediately after the burn is explained by the presence of various ions in the ash (Pino-del et al., 2008; Úbeda et al., 2009; Pereira et al., 2011; Campos et al., 2016). The high pH value of ash 1 sample, which includes a high carbonate content, and the grey colour suggests that this ash was produced at high temperature with high fire severity (Etiegni and Campbell, 1991; Ulery and Graham, 1993; Henig-Sever et al., 2001; Goforth et al., 2005; Pereira et al., 2008). The organic matter levels vary from 1.1% to 39.8% and the carbonate values range from 10.4% to 37.8%. The extremely high OM content (39.8%) in ash 2 together with the high values in ash 3 (18.9%) and ash 4 (17.8%) coupled with their dark colours suggests the incomplete combustion of vegetation (Goforth et al., 2005; Bodí et al., 2014).

Table 1
pH, EC, OM and carbonate content observed in ash samples, including colour.

Sample	pH	EC (μS cm^{-1})	OM (%)	Carbonate (%)	Colour
Ash 1	11.6	1173	1.11	37.77	Grey
Ash 2	9.1	28,500	39.83	10.37	Light grey
Ash 3	9.7	2600	18.89	19.28	Black
Ash 4	9.6	18,900	17.77	17.13	Black

3.4. Potentially toxic metals (PTMs) in ash

A study of ash is significant in understanding the PTM mobility in fire-affected landscapes, as ash often contains elevated concentration of PTMs (Plumlee et al., 2007; Pereira and Ubeda, 2010; Odigie and Flegal, 2011, 2014). Though light grey, and grey coloured ash were present in the study area, most of the area was covered with black ash, consistent with incomplete combustion of vegetation. Due to the increase in areal coverage, black ash may have significantly influenced the soil characteristics and the PTM concentrations in the soil. The PTM concentrations in ash and the descriptive statistics are given in Table 2.

The ash residue after the controlled burn revealed variation in PTM concentrations compared to PTM concentrations in the immediate post-burn soil environment. Ash sample 1 is sourced from the burning of a fallen tree trunks whereas, other samples are mostly attributed to being formed from bush and canopy leaf litter. Based on the PTM analysis data, it was found that grey coloured ash displayed the lowest Mn and Ni values, and the light grey ash displayed the second lowest. The Mn and Ni values were highest in the black ash attributes to the increase in concentrations of Mn and Ni, in incomplete combustion ash (black ash), most probably related to burn intensity. Conversely Pb and As shown higher concentration in the grey ash when compared to the black ash. Among the nine PTMs, Mn (1270 mg kg^{-1}) and Zn (474 mg kg^{-1}) achieved the highest and second highest median concentrations in ash respectively, whereas, Cd (0.06 mg kg^{-1}) represented the lowest concentration. All these values correspond with PTM concentrations in the immediate post-burn soil environment.

The median concentration of Mn, Zn, Cu, Ni and Co in ash is found to be higher than the concentration in the surrounding immediate post-burn soil environment, whereas the concentration of As, Cd, Cr and Pb are found to be higher in soil than in ash. Relative increases in concentration of Cu (56.1 to 207 mg kg^{-1}) and Mn (329 to 1270 mg kg^{-1}) found in ash from this study is of concern as ash represents a significant vector for PTM redistribution in the environment. The varying concentrations of PTMs in ash depends on a number of factors such as pre-burn metal concentration in the soil due to legacy mining activities, percentage SOM, age of the litter level, types of vegetation and vegetation density in the forest, the nature of materials burned (such as barks, leaves and litter), duration of the fire, maximum temperature reached, soil types, moisture content in the soil, post-burn meteorological conditions and many other local factors (Someshwar, 1996; Pereira et al., 2009; Pereira and Ubeda, 2010; Machado et al., 2015). Plants in the study area usually absorb PTMs from soil and the PTM

Table 2
PTM concentrations in ash samples include descriptive statistics.

Sample ID	As	Cd	Co	Cr	Cu	Mn	Ni	Pb	Zn	Colour
Ash 1	260.0	0.22	9.1	9.0	56.1	330.0	9.5	14.9	221.0	Grey
Ash 2	24.0	0.08	5.7	8.0	147.0	980.0	19.1	6.0	555.0	Light grey
Ash 3	21.7	0.02	12.6	5.0	142.5	2790.0	37.6	3.6	527.0	Black
Ash 4	20.8	0.03	9.0	9.0	207.0	1560.0	35.9	1.8	422.0	Black
Median	22.9	0.06	9.1	8.5	144.8	1270	27.5	4.8	474.5	
Range	239.2	0.2	6.9	4	150.9	2460	28.1	13.1	334	
IQR	180	0.16	5.2	3.3	114.3	1990	25.3	10.4	276.8	

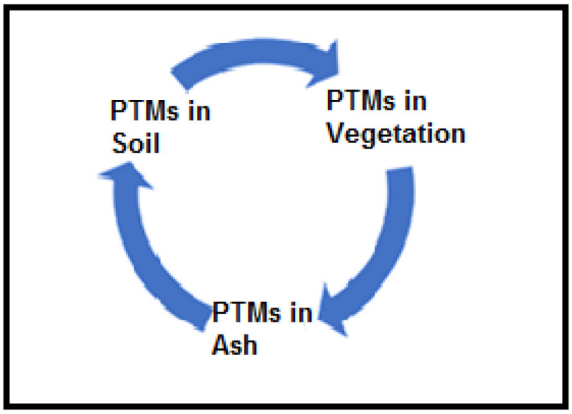

Fig. 5. Showing the mobilization of PTMs in the forest ecosystem during fire.

concentration in the vegetation particularly depends on the PTM levels in the soil and the age of the vegetation. During controlled burn, the PTM in the vegetation comes back to the soil through ash, which may result in a significant increase in PTM concentration in soil and it can be considered as a cyclic concentration process (Fig. 5).

Previous studies revealed the presence of PTMs (Mn, Zn, Cu, Pb, Hg, Cd and As) in ash samples found in forests across the globe (Ferreira et al., 2005; Plumlee et al., 2007; Pereira and Ubeda, 2010; Bogacz et al., 2011; Odigie and Flegal, 2011, 2014; Campos et al., 2016). To illustrate this, Plumlee et al. (2007) reported significantly elevated concentration of PTMs (As (up to 140 mg kg^{-1}), Pb (up to 344 mg kg^{-1}), Cu (1370 mg kg^{-1}) and Zn (up to 2800 mg kg^{-1})) in ash samples after the southern California forest fire in USA, and these results persisted for a year following the fire. Similarly, distribution of Co (3–11 mg kg^{-1}), Cu (15–69 mg kg^{-1}), Ni (6–15 mg kg^{-1}), Pb (7–42 mg kg^{-1}) and Zn (65–500 mg kg^{-1}) were observed in ash after the 2012 Williams fire in Los Angeles, USA (Odigie and Flegal, 2014) and Kristensen et al. (2014) reported mobilization of natural and industrial Pb in ash samples after three large uncontrolled intense wildfire events in 2011 in Australia. Many of the PTM mobilised studies highlight Mn as being present in the highest concentration (Parra et al., 1996; Costa et al., 2014; Campos et al., 2016). The broad ranges of PTM concentrations found in Maldon Study area reflect the heterogeneous nature of ash formed in a historical mining site, probably due to the variation in plant species and parts burned, a mixture of the flammable materials and their original chemistry, burn intensity, and combustion completeness (Someshwar, 1996; Bodí et al., 2014). Thus, in general forest fires, including controlled burns can be considered as a vector for environmental alterations that may influence human and ecosystems health due to mobilization on PTM.

3.5. *Effects of controlled burn on PTMs in soil and post-burn temporal variations*

The concentration of PTMs (As, Cd, Co, Cr, Cu, Mn, Ni, Pb, and Zn) in the soils of the study area (Maldon legacy mine site) two days before the controlled burn (2DBB), two days after burn (2DAB), at the end of 03/06/09/12 months (3MAB/6 MAB/9MAB/12MAB) and immediately after the major rainfall (IAMR) event(s) in the post-burn environment

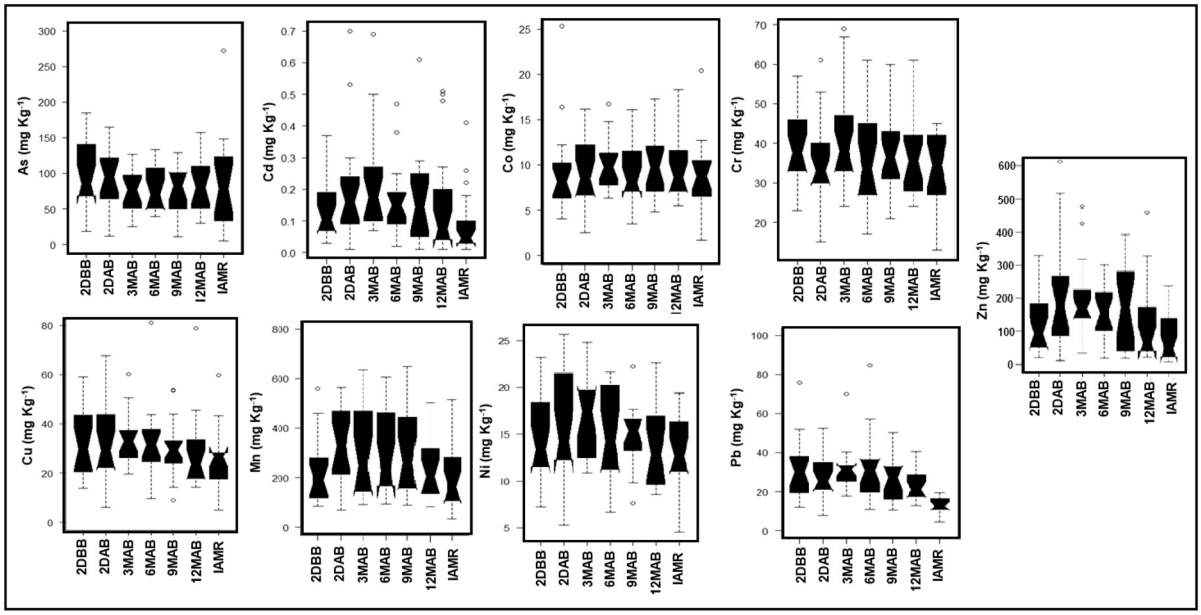

Fig. 6. Concentrations of PTMs during various sampling period (2DBB- two days before burn, 2DAB- two days after burn, 3MAB- three months after burn, 6MAB- six months after burn, 9MAB- nine months after burn, 12MAB-twelve months after burn and IAMR- immediately after major rainfall) (n = 18).

are depicted in Fig. 6. The median concentrations of PTMs before burn was in the order: Mn > Zn > As > Cr > Cr > Pb > Ni > Co > Cd. When comparing the BB (before burn) PTM concentrations with Australian and Victorian top soil averages (de Caritat and Cooper, 2016), As, Pb, Cu and Zn exceeded the averages, mostly attributable legacy mining activities. When comparing with ecological investigation levels (EILs) set by the Department of Environment and Conservation (DEC) of the State of Western Australia, As exceeded the EIL and are considered a risk to the human and ecosystems health due to their increase in concentration and toxicity (DEC, 2010).

The median concentrations of Zn, Mn, Cd, As, Co, and Ni in the soil immediately after the burn were 1.9, 1.7, 1.5, 1.1, 1.1 and 1.1 times higher (respectively) than their levels in the pre-burn soil, with statistical significance ($p < 0.05$) observed only in Mn and Zn concentrations. By contrast, Cr and Cu concentrations varied by a factor of 0.9 and Pb concentrations varied by a factor of 0.8. The maximum increases observed in the study area were As (28.9), Zn (9.7), Mn (9.3), Cd (5.9), Cu (3.7) and Ni (2.7) times the pre-burn concentrations.

Of the PTMs considered, Zn (87%) and Mn (72%) record the highest and second highest increase in median values in the study area in the immediate post-burn soil environment, followed by Cd (45%), and As (11%). Other PTM variations are < 10% and are thus considered not to be significant. All the PTMs, except Cr, Cu and Pb showed an increase in concentration immediately after the controlled burn. The PTMs mobilization observed in this study is consistent with other studies, which report toxic metal mobilization after wild fires (Ignatavièius et al., 2006; Plumlee et al., 2007; Bogacz et al., 2011; Odigie and Flegal, 2011; Norouzi and Ramezanpour, 2013; Costa et al., 2014; Kristensen et al., 2014; Odigie and Flegal, 2014; Campos et al., 2016).

For example, Parra et al. (1996) noted a 100% increase in Mn concentrations and Costa et al. (2014) noted a five times Mn enrichment in ash relative to the surrounding soil after forest fires, whereas Chambers and Attiwill (1994) observed 279% increase in water soluble Mn concentrations after a laboratory heating experiment. The increase in Mn is attributed to a significant contribution to the soil from ash (in the form of oxides) from burned vegetation, as Mn is known to be concentrated in tree leaves compared to bark (Kabata-Pendias, 2011). Some results of this study support the above postulate. To illustrate this, ash sample 1, formed by the burning of a tree trunk showed only 330 mg kg^{-1} of Mn, whereas ash sample 4, formed by the burning of tree leaves and bush (mostly) showed a high Mn concentration of 2790 mg kg^{-1}.

Although the concentration of Cd is less (compared to other metals) in the soil, it showed a 45% increase immediately after the controlled burn. Existing studies revealed that increase in Cd concentration in soil in the immediate post-burn environment is due to sorption by organic matter, coupled with the association of Mn oxides, Fe and Al hydroxides and clay mineral fixation (Sipos et al., 2005; Kabata-Pendias, 2011). Cu showed no change in concentration in the immediate post-burn environment (from 30 to 29 mg kg^{-1} only), but showed higher concentrations (145 mg kg^{-1}) in ash, consistent with other studies (Costa et al., 2014; Campos et al., 2016). The results clearly illustrate that PTM mobility in the soil environment during and after the controlled burn. The increase in concentration is considered to be due to the accumulation and dispersion of ash which contains elevated PTM concentrations (Plumlee et al., 2007; Costa et al., 2014; Campos et al., 2016), while the decrease may be due to the volatilization of metals during burning (Biswas et al., 2007; Campos et al., 2015; Abraham et al., 2018). The PTM mobility or change characteristics depends on a number of factors including accumulation and deposition of PTMs in the soil, type and density of the vegetation in the area, duration of the fire, maximum temperature reached and many other local factors (Pereira et al., 2011). Thus, further studies are needed to establish the relative influence of these factors in mining affected landscapes.

A knowledge of the detailed PTM dynamics in the environment is essential for ecosystem management and sustainability (Campos et al., 2016). The concentrations of PTMs in the post-burn environment during all the sampling periods are given in Fig. 6 and the descriptive statistics are depicted in Table S2 (Supplementary table). Among the nine PTMs considered in the study, As, Cd, Co, Mn, Ni and Zn showed increase in median concentration (only Mn and Zn are statistically significant) two days after burn (2DAB), however, Cr and Pb showed a reduction in concentration, with no variation in the case of Cu.

Among the first group of PTMs, Mn and Zn evidenced approximately similar distribution patterns in the post-burn environment. Two days after burn, both have increased their concentrations compared to pre-burn concentrations and showed a reduction in the succeeding sampling periods, except 9MAB sampling, were both showed a small increase. The reduction was in such a way that 12MAB, Zn falls below pre-burn levels, whereas Mn falls below its pre-burn level only after the major rainfall event. Two other PTMs, Cd and Ni, demonstrated similar kinds of post-burn temporal distribution patterns. Two days after burn, both displayed increase in concentrations, and the increase continued till 3MAB. Then the levels started decreasing (except in 9MAB samples, where Cd level is same as previous level and Ni showed negligible increase), and after the major rainfall event, both levels reached or reduced below pre-burn levels. Cobalt also showed a similar pattern to Cd and Ni, except no change was observed after the major rainfall. The increase in concentration of Cd, Ni and Co 3MAB samples is attributed to the continued mixing of ash with the soil. In the case of As, it showed an increase in median concentration immediately after the burn, which was reduced in 3MAB and 6MAB, but further increases were observed in 9MAB followed by a slight reduction in 12MAB and IAMR samples.

Unlike other six PTMs, the concentrations of Cr and Pb decreased two days after the controlled burn, but both showed an increase 3MAB. In the following sampling period, there was reduction or 'no change' in both the cases, except an increase in Cr concentration 9MAB samples, compared to previous (6MAB) samples.

Even though, a number of minor rainfall events occurred before September 2016 in the post-burn environment, they were not substantial enough to remobilize ash and surface soils. These less intense rainfall events, specifically rainfall occurred 6MAB sampling may have mobilised ash and ash enriched surface soils, including the associated contaminants within the study area itself resulting in the increase in median concentrations of some of the PTMs in 9MAB samples (Shakesby, 2011; Moody et al., 2013; Devi and Yadav, 2018). The rainfall during the second week of Sep 2016 was so intense that > 155 mm rainfall occurred within a fortnight. The subsequent erosion was so severe that it deepened and widened the existing creek in many places and flooded number of locations in Central Victoria. This intense rainfall may have transported much of the ash-soil mixture away from the study area, resulting in the observed significant decreases in PTM concentrations immediately after the rainfall event, consistent with other studies (Gallaher and Koch, 2004; Businelli et al., 2009; Burke et al., 2010; Violante et al., 2010; Stein et al., 2012; Prats et al., 2012, 2014; Machado et al., 2015). In support of this postulation, Costa et al. (2014) noted the decrease in concentration of Mn in soil and the subsequent increase in the downstream water environment eight and 12 months after a fire. Conversely, the slight increase in concentration of Cu in the study area after the rainfall is consistent with the reduced solubility and reduced mobility of Cu compounds than other PTM compounds (Wuana and Okieimen, 2011; Zhang et al., 2012; Devi and Yadav, 2018). In addition to this lateral mobilization, some of the PTMs may be leached into subsurface soil layers, thus influencing groundwater, especially when the groundwater levels are close to the surface (Verma et al., 2016; Tahiri et al., 2017; Vink et al., 2017). This potential impact required further investigation, specifically where frequent controlled burns practicing and the groundwater is supply as potable water resources.

3.6. Relationship between soil physicochemical characteristics and PTMs concentrations

PTMs in the soils of legacy mine sites generally have complex chemical and physical relationships with soil characteristics (pH, EC and SOM), and it is known that numerous factors control their relative abundance (Krishna et al., 2013). This includes, but is not limited to, PTM concentration in the parent rock or material, extent of weathering and soil formation, anthropogenic activities include intensity and types of mining, duration of the mining, types of metal processing and distance to the processing site, mode of transport to the processing site, topography of the area, and climate (Wei and Yang, 2010; Krishna et al., 2013). This inhomogeneity is visible in the study area.

Many studies have revealed that the physicochemical characteristics of soil, specifically pH and SOM played a significant role in the retention and mobilization of PTMs in the soil (Chlopecka et al., 1996; Certini, 2005; Zhang et al., 2012; Mehes-Smith et al., 2013). We applied the Spearman Rank correlation in R (version 3.4.1) to investigate the relationship between physicochemical characteristics (pH, EC, SOM and carbonate) of the soil and the PTM in the soil in four sampling periods. The correlation coefficient of before burn (2DBB) and immediately after burn (2DAB) samples and before major rainfall (12MAB) and immediately after major rainfall (IAMR) samples are depicted in Tables 3 and 4 respectively. In both the pre-burn and immediate post-burn environment, statistically strong (0.6–0.79) correlation are observed between the physicochemical properties and PTMs for pH with Ni ($r = 0.62$, $p < 0.01$) and Co ($r = 0.60$, $p < 0.01$) (both environment), and the PTMs themselves (Table 3). Similar correlation between pH with Ni and Co has been reported by Wilde et al. (2004). Apart from this, other observed moderate positive linear correlations are reported for Cr ($r = 0.54$, $p < 0.05$), SOM with Cd ($r = 0.53$, $p < 0.05$), Cu ($r = 0.51$, $p < 0.05$) and Ni ($r = 0.50$, $p < 0.05$) and carbonate with Cr ($r = 0.57$, $p < 0.05$) and Ni ($r = 0.50$, $p < 0.05$) in both the pre-burn and immediate post-burn environment. For the other PTMs, these correlations are not statistically significant with physicochemical characteristics. The very strong and strong correlations observed between the PTMs themselves (Cd with Zn, Cu with Pb and Zn; etc.) highlight their common origin and behaviour patterns. The strong correlation between Pb with As ($r = 0.77$, $p < 0.01$) and Zn with As ($r = 0.76$, $p < 0.01$) suggest a common origin that being the mineralized Ordovician sedimentary rock with galena (PbS) and sphalerite (ZnS) as potential primary sulphides. The very strong correlation between As and Mn ($r = 0.80$, $p < 0.01$) determined that Mn is a significant metal in the soil, and may control arsenate (major form of As) mobility, also reported by Sultan (2007).

The correlation coefficient of data collected from 12MAB samples are significant as the samples collected few days before the occurrence of major rainfall. Here, pH has a strong correlation with Mn ($r = 0.71$, $p < 0.01$), Zn ($r = 0.67$, $p < 0.01$), and Cu ($r = 0.63$, $p < 0.01$) and carbonate shown very strong correlation with Ni ($r = 0.87$, $p < 0.01$), Cu ($r = 0.82$, $p < 0.01$), and Zn ($r = 0.81$, $p < 0.01$), and strong correlation with Co ($r = 0.77$, $p < 0.01$), Pb ($r = 0.76$, $p < 0.01$), Cd ($r = 0.68$, $p < 0.01$) and Mn ($r = 0.62$, $p < 0.01$). EC displayed correlation only with Cd ($r = 0.58$, $p < 0.05$) and is a moderate one. The correlation between the physicochemical characteristics and PTMs has altered immediately after the major rainfall due to the influence of the rainfall on surface soil. For example, pH has a strong correlation with Cu ($r = 0.69$, $p < 0.01$), Zn ($r = 0.67$, $p < 0.01$) and Ni ($r = 0.60$, $p < 0.01$) only, not with Mn as in the pre-rainfall environment. The relationship of EC and SOM with PTMs has increased after the major rainfall. To illustrate this: (i) EC highlighted very strong correlation with number of PTMs such as Cd ($r = 0.91$, $p < 0.01$), Mn ($r = 0.89$, $p < 0.01$), and Zn ($r = 0.83$, $p < 0.01$), and strong correlation with Cu ($r = 0.75$, $p < 0.01$), Pb ($r = 0.67$, $p < 0.01$) and Ni ($r = 0.60$, $p < 0.01$). (ii) SOM shown significant strong correlation with Cd ($r = 0.75$, $p < 0.01$), Mn ($r = 0.74$, $p < 0.01$) and Zn ($r = 0.71$, $p < 0.01$). However, the relationship of carbonate with PTMs decreased in the immediate post-rainfall environment. For example, carbonate shown strong correlation only with Co ($r = 0.64$, $p < 0.01$), unlike the previous environment.

Previous studies noted the relationship between soil physicochemical properties and PTM concentrations, including their retention and mobility in the soil (Chlopecka et al., 1996; Mehes-Smith et al., 2013). However, in this study there is limited strong linear positive correlation observed between the physicochemical characteristics and PTMs in the pre-burn and immediate post-burn environment, and this warrants more investigation. However, the relationship changed after one year, in the pre-rainfall and immediate post-rainfall environment. Moreover, although the correlation between the physicochemical characteristics of the soil and PTMs in the soil are similar in the pre-burn and immediate post-burn environment, it is considerably inhomogeneous when we compare pre-rainfall and immediate post-rainfall environment. This suggest that controlled burn has not influenced the relationship, whereas, strong a rainfall event significantly influenced the soil physicochemical and PTM concentration relationship.

4. Conclusion

This study investigates the remobilization of potentially toxic metals (PTMs) in the surface soils of a legacy mine site in the Maldon area of Central Victoria, Australia after a controlled burn to assess their temporal concentration variations. Surface soil samples were collected two days before, two days after the controlled burn and five more times in the succeeding post-fire environment with three months interval, and

Table 3
Spearman rank correlation coefficient between the physicochemical properties of soil and PTMs in the pre-burn (above diagonal elements) and the post-burn (below the diagonal elements) environment (number of samples = 18).

	pH	EC	SOM	Carb	As	Cd	Co	Cr	Cu	Mn	Ni	Pb	Zn
pH		−0.48	0.43	0.04	0.32	0.37	***0.61****	**0.54***	0.38	0.35	***0.62****	−0.04	*0.46*
EC	−0.48		0.28	0.37	−0.23	−0.10	−0.03	−0.02	0.14	0.04	−0.08	−0.03	−0.21
SOM	0.43	0.28		0.37	0.10	**0.53***	0.38	0.44	**0.51***	0.36	**0.50***	0.28	0.49
Carb	0.04	0.38	*0.37*		0.32	0.43	0.46	**0.57***	0.48	0.45	**0.50***	0.48	0.40
As	0.32	−0.23	0.10	0.32		***0.75****	***0.65****	0.48	***0.60****	***0.80****	**0.53***	***0.77****	***0.76****
Cd	0.37	−0.10	**0.53***	0.43	***0.74****		*0.70**	*0.66**	*0.72**	***0.73****	***0.75****	***0.75****	***0.98****
Co	***0.61****	−0.03	0.38	0.46	***0.65****	***0.70****		*0.86**	0.76**	***0.73****	***0.94****	0.30	***0.76****
Cr	**0.54***	−0.02	0.44	**0.57***	0.48	***0.66****	***0.86****		*0.77**	0.47	***0.95****	0.32	***0.72****
Cu	0.38	*0.14*	**0.51***	*0.48*	***0.60****	***0.72****	***0.76****	***0.77****		**0.58***	***0.79****	0.46	***0.74****
Mn	0.35	0.04	0.36	0.45	***0.80****	***0.73****	***0.73****	0.47	**0.57***		***0.61****	***0.62****	***0.71****
Ni	***0.62****	−0.08	**0.50***	**0.50***	**0.53***	***0.75****	***0.94****	***0.95****	***0.79****	***0.61****		0.32	***0.82****
Pb	−0.04	−0.03	0.28	0.48	***0.77****	***0.75****	*0.30*	0.32	0.46	***0.62****	0.32		***0.67****
Zn	0.46	−0.21	0.49	0.40	***0.76****	***0.98****	***0.76****	***0.72****	***0.74****	***0.71****	***0.81****	***0.67****	

NB: **Significant positive correlations ($p < 0.01$, ***bold italics***; *$p < 0.05$, **bold**) are highlighted.

Table 4
Spearman rank correlation coefficient between the physicochemical properties of soil and PTMs in the pre-rainfall (12MAB) (above diagonal elements) and the post-rainfall (IAMR) (below the diagonal elements) environment(s) (number of samples = 18).

	pH	EC	SOM	Carb	As	Cd	Co	Cr	Cu	Mn	Ni	Pb	Zn
pH		−0.04	−0.11	***0.55****	***0.53****	0.49	***0.54****	0.48	***0.63*****	***0.71*****	***0.54****	***0.51****	***0.67*****
EC	0.42		0.31	***0.56****	0.19	***0.58****	0.32	0.06	0.48	0.45	0.40	0.44	0.46
SOM	0.34	***0.89*****		0.18	0.18	0.33	−0.21	0.30	0.13	0.09	0.40	0.28	0.22
Carb	0.38	***0.53****	***0.68*****		0.28	***0.68*****	***0.77*****	***0.55****	***0.82*****	***0.62*****	0.***87*****	***0.76*****	***0.81*****
As	0.48	0.43	0.19	0.18		0.52	0.28	0.04	***0.61*****	***0.56****	0.28	0.47	***0.52****
Cd	***0.54****	***0.91*****	***0.75*****	0.49	***0.62*****		***0.53****	0.35	***0.78*****	***0.84*****	***0.70*****	***0.85*****	***0.90*****
Co	0.48	0.***54****	0.43	***0.64*****	***0.72*****	***0.66*****		***0.54****	***0.80*****	***0.61****	***0.85*****	***0.64*****	***0.67*****
Cr	0.24	0.37	0.42	***0.58****	0.45	***0.53****	***0.66*****		***0.52****	0.30	***0.69****	0.47	0.43
Cu	***0.69*****	***0.75*****	***0.57****	0.49	***0.79*****	***0.88*****	***0.81*****	***0.54****		***0.77*****	***0.83*****	***0.88*****	***0.83*****
Mn	***0.59****	***0.89*****	***0.74*****	0.45	***0.65*****	***0.90*****	***0.67*****	0.44	***0.88*****		***0.60*****	***0.84*****	***0.84*****
Ni	***0.60*****	***0.60*****	***0.52****	***0.58****	***0.66*****	***0.79*****	***0.85*****	***0.74*****	***0.86*****	***0.71*****		***0.67*****	***0.83*****
Pb	0.31	***0.67*****	***0.54****	0.43	***0.71*****	***0.83*****	***0.69*****	***0.63*****	***0.76*****	***0.75*****	***0.78*****		0.***88*****
Zn	***0.67*****	***0.83*****	***0.71*****	***0.58****	***0.64*****	***0.93*****	***0.72*****	***0.63*****	***0.92*****	***0.89*****	***0.88*****	***0.83*****	

NB: **Significant positive correlations ($p < 0.01$, ***bold italics***; *$p < 0.05$, **bold**) are highlighted.

analysed for PTM concentrations and physicochemical characteristics.

The results revealed the presence of PTM mobilization after the controlled burn and most of the PTMs (As, Cd, Cu, Mn, Ni, and Zn) evidenced an increase immediately after the burn but a reduction in the subsequent post-burn environment. The increase is postulated to be associated with addition of PTM enriched ash to the soil, while the decrease is due to the removal of ash and surface soil by rainfall runoff. The study also highlighted the increase in pH, EC and SOM (%) immediately after the fire, which subsequently reduced in the post-burn environment. The increase in soil pH and EC immediately after fire is attributed to the presence of various alkaline oxides and hydroxides derived from ash. The increase in SOM is due to the presence of pyrogenic carbon compounds in the ash resulting from incomplete combustion. Unlike other physicochemical characteristics, carbonate showed little variation in concentration, potentially due to insufficient temperature during fire. In general, controlled burn has been found to alter the PTMs concentrations in the soil by combustion, heating and ash deposition.

The controlled burn or prescribed fire land management strategy may result in the remobilization of considerable amounts of PTMs, which may create significant offsite environmental impacts. Notwithstanding the relatively small study area related to this investigation the results have global ramifications as more than a million legacy mine sites exist across the globe, many of which are in fire-prone forest areas. Moreover, climate change and the resulting projected increased in forest fire frequency highlights the environmental significance of increased PTM mobilization. Hence, the practice of controlled burning should be carefully considered as a wildfire mitigation and land management strategy, and consideration must be given to avoid mobilization of metals in legacy mining areas and indeed other areas where PTM contamination is reported.

Conflict of interest

Authors declared that there is no conflict of interest.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.geoderma.2018.06.010.

References

Abraham, J., Dowling, K., Florentine, S., 2017a. The unquantified risk of post-fire metal concentration in soil: a review. Water Air Soil Pollut. 228 (5), 175. http://dx.doi.org/10.1007/s11270-017-3338-0.

Abraham, J., Dowling, K., Florentine, S., 2017b. Risk of post-fire metal mobilization into surface water resources: a review. Sci. Total Environ. 599–600, 1740–1755. http://dx.doi.org/10.1016/j.scitotenv.2017.05.096.

Abraham, J., Dowling, K., Florentine, S.K., 2018. Effects of prescribed fire and post-fire rainfall on mercury mobilization and subsequent contamination assessment in a legacy mine site in Victoria, Australia. Chemosphere 190, 144–153. http://dx.doi.org/10.1016/j.chemosphere.2017.09.117.

Adriano, D.C., 2001. Trace elements in Terrestrial Environments: Biogeochemistry, Bioavailability and Risks of Metals, II ed. Springer - Verlag, New York.

Alcañiz, M., Outeiro, L., Francos, M., Úbeda, X., 2018. Effects of prescribed fires on soil properties: a review. Sci. Total Environ. 613, 944–957. http://dx.doi.org/10.1016/j.scitotenv.2017.09.144.

Armienta, M.A., Mugica, V., Resendiz, I., Arzaluz, M.G., 2016. Arsenic and metal mobility in soils impacted by tailings at Zimapan Mexico. J. Soils Sediments 16 (4), 1267–1278. http://dx.doi.org/10.1007/s1136.

Arocena, J., Opio, C., 2003. Prescribed fire-induced changes in properties of sub-boreal forest soils. Geoderma 113 (1), 1–16. http://dx.doi.org/10.1016/S0016-7061(02)00312-9.

Augustine, D.J., Brewer, P., Blumenthal, D.M., Derner, J.D., von Fischer, J.C., 2014. Prescribed fire, soil inorganic nitrogen dynamics, and plant responses in a semiarid grassland. J. Arid Environ. 104, 59–66. http://dx.doi.org/10.1016/j.jaridenv.2014.01.022.

Badía, D., Marti, C., 2003. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Res. Manag. 17 (1), 23–41. http://dx.doi.org/10.1080/15324980301595.

Beganyi, S.R., Batzer, D.P., 2011. Wildfire induced changes in aquatic invertebrate communities and mercury bioaccumulation in the Okefenokee Swamp. Hydrobiologia 669 (1), 237–247. http://dx.doi.org/10.1007/s10750-011-0694-4.

Bennett, L.T., Aponte, C., Baker, T.G., Tolhurst, K.G., 2014. Evaluating long-term effects of prescribed fire regimes on carbon stocks in a temperate eucalypt forest. For. Ecol. Manag. 328, 219–228. http://dx.doi.org/10.1016/j.foreco.2014.05.028.

Bento-Gonçalves, A., Vieira, A., Úbeda, X., Martin, D., 2012. Fire and soils: key concepts and recent advances. Geoderma 191, 3–13. http://dx.doi.org/10.1016/j.geoderma.2012.01.004.

Biswas, A., Blum, J.D., Klaue, B., Keeler, G.J., 2007. Release of mercury from Rocky Mountain forest fires. Glob. Biogeochem. Cycles 21 (1). http://dx.doi.org/10.1029/2006 GB002696.

Bodí, M.B., Martin, D.A., Balfour, V.N., Santín, C., Doerr, S.H., Pereira, P., Santin, C., Mataix-Solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 130, 103–127. http://dx.doi.org/10.1016/j.earscirev.2013.12.007.

Bogacz, A., Wozniczka, P., Labaz, B., 2011. Concentration and pools of heavy metals in organic soils in post-fire areas used as forests and meadows. J. Elem. 16 (4). http://dx.doi.org/10.5601/jelem.2011.16.4.01.

BoM, 2018. Bureau of Meteorology - Rainfall Data. http://www.bom.gov.au/climate/data/, Accessed date: 12 February 2018.

Burke, M.P., Hogue, T.S., Ferreira, M., Mendez, C.B., Navarro, B., Lopez, S., Jay, J.A., 2010. The effect of wildfire on soil mercury concentrations in Southern California watersheds. Water Air Soil Pollut. 212 (1–4), 369–385. http://dx.doi.org/10.1007/s11270-010-0351-y.

Burke, M.P., Hogue, T.S., Barco, J., Wessel, C., Kinoshita, A.Y., Stein, E.D., 2013. Pre- and post-fire pollutant loads in an urban fringe watershed in Southern California. Environ. Monit. Assess. 185, 10131–10145. http://dx.doi.org/10.1007/s10661-013-3318-9.

Burton, C.A., Hoefen, T.M., Plumlee, G.S., Baumberger, K.L., Backlin, A.R., Gallegos, E., Fisher, R.N., 2016. Trace elements in stormflow, ash, and burned soil following the 2009 station fire in Southern California. PLoS One 11 (5), e0153372. http://dx.doi.org/10.10371/journal.pone.0153372.

Businelli, D., Massaccesi, L., Onofri, A., 2009. Evaluation of Pb and Ni mobility to groundwater in calcareous urban soils of Ancona, Italy. Water Air Soil Pollut. 201 (1–4), 185–193. http://dx.doi.org/10.1007/s11270-008-9936-0.

Campos, I., Vale, C., Abrantes, N., Keizer, J.J., Pereira, P., 2015. Effects of wildfire on mercury mobilisation in eucalypt and pine forests. Catena 131, 149–159. http://dx.doi.org/10.1016/j.catena.2015.02.024.

Campos, I., Abrantes, N., Keizer, J.J., Vale, C., Pereira, P., 2016. Major and trace elements in soils and ashes of eucalypt and pine forest plantations in Portugal following a wildfire. Sci. Total Environ. 572, 1363–1376. http://dx.doi.org/10.1016/j.scitotenv.2016.01.190.

de Caritat, P., Cooper, M., 2016. A continental-scale geochemical atlas for resource exploration and environmental management: the National Geochemical Survey of Australia. Geochem. Explor. Environ. Anal. 16, 3–13. http://dx.doi.org/10.1144/geochem2014-322.

Castellinou, M., Kraus, D., Miralles, M., 2010. Prescribed burning and suppression fire techniques: from fuel to landscape management. In: Paper Presented at the Catalonian Programme of Fire Management, GRAF Team Actions, Spain.

Certini, G., 2005. Effects of fire on properties of forest soils: a review. Oecologia 143 (1), 1–10. http://dx.doi.org/10.1007/s00442-004-1788-8.

Chambers, D., Attiwill, P., 1994. The ash-bed effect in *Eucalyptus regnans* forest: chemical, physical and microbiological changes in soil after heating or partial sterilisation. Aust. J. Bot. 42 (6), 739–749. http://dx.doi.org/10.1071/BT9940739.

Cherry, D.P., Wilkinson, H.E., 1994. Bendigo and Part of Mitiamo, 1:100,000 Map Geological Report, Melbourne.

Chlopecka, A., Bacon, J., Wilson, M., Kay, J., 1996. Forms of cadmium, lead, and zinc in contaminated soils from Southwest Poland. J. Environ. Qual. 25 (1), 69–79. http://dx.doi.org/10.2134/jeq1996.00472425002500010009x.

Clark, M., Walsh, S., Smith, J., 2001. The distribution of heavy metals in an abandoned mining area; a case study of Strauss Pit, the Drake mining area, Australia: implications for the environmental management of mine sites. Environ. Geol. 40 (6), 655–663. http://dx.doi.org/10.1007/s002549900073.

Cobbina, S., Myilla, M., Michael, K., 2013. Small scale gold mining and heavy metal pollution: assessment of drinking water sources in Datuku in the Talensi-Nabdam District. Int. J. Sci. Technol. Res. 2, 96–100.

Costa, M.R., Calvão, A.R., Aranha, J., 2014. Linking wildfire effects on soil and water chemistry of the Marão River watershed, Portugal, and biomass changes detected from Landsat imagery. Appl. Geochem. 44, 93–102. http://dx.doi.org/10.1016/j.apgeochem.2013.09.009.

DEC (Department of Environment and Conservation, Western Australia), 2010. Contaminated Sites Management Series - Assessment levels for Soil, Sediment and Water. Version 4, Revision 1. 53 p. Available at. www.dec.wa.gov.au/contaminatedsites (accessed 20 Jul 2017).

Devi, N.L., Yadav, I.C., 2018. Chemometric evaluation of heavy metal pollutions in Patna region of the Ganges alluvial plain, India: implication for source apportionment and health risk assessment. Environ. Geochem. Health 1–16. http://dx.doi.org/10.1007/s10653-018-0101-4.

Egold, 2018. Mining Technology Overview, A Nation's Heritage. Retrieved from. http://www.egold.net.au/biogs/EG00009b.htm, Accessed date: 28 March 2018.

Erickson, R.J., Nichols, J.W., Cook, P.M., Ankley, G.T., 2008. Bioavailability of chemicals in the aquatic environment. In: Di Giulio, R., Hinton, D. (Eds.), The Toxicology of Fishes. Taylor & Francis, New York, NY, USA, pp. 9–54.

Esshaimi, M., Quazzani, N., Avila, M., Perez, G., Valiente, M., Mandi, L., 2012. Heavy metal contamination of soils and water resources Kettara abandoned mine. Am. J. Environ. Sci. 8 (3), 253–261. http://dx.doi.org/10.3844/ajessp.2012.253.261.

Etiegni, L., Campbell, A., 1991. Physical and chemical characteristics of wood ash. Bioresour. Technol. 37 (2), 173–178. http://dx.doi.org/10.1016/0960-8524(91)90207-Z.

Faria, S.R., De La Rosa, J., Knicker, H., González-Pérez, J.A., Villaverde, J., Keizer, J.J., 2015. Wildfire-induced alterations of topsoil organic matter and their recovery in Mediterranean eucalypt stands detected with biogeochemical markers. Eur. J. Soil Sci. 66 (4), 699–713. http://dx.doi.org/10.1111/ejss.12254.

Fernandes, P.M., Davies, G.M., Ascoli, D., Fernández, C., Moreira, F., Rigolot, E., ... Molina, D., 2013. Prescribed burning in southern Europe: developing fire management in a dynamic landscape. Front. Ecol. Environ. 11 (s1). http://dx.doi.org/10.1890/120298.

Fernandez, P.M., Botelho, H.S., 2003. A review of prescribed burning effectiveness in fire hazard reduction. Int. J. Wildland Fire 12, 117–128. http://dx.doi.org/10.1071/WF02042.

Ferreira, A., Coelho, C., Boulet, A., Lopes, F., 2005. Temporal patterns of solute loss following wildfires in Central Portugal. Int. J. Wildland Fire 14 (4), 401–412. http://dx.doi.org/10.1071/WF05043.

Gabet, E.J., Bookter, A., 2011. Physical, chemical and hydrological properties of Ponderosa pine ash. Int. J. Wildland Fire 20 (3), 443–452. http://dx.doi.org/10.1071/WF09105.

Gallaher, B.M., Koch, R.J., 2004. Cerro Grande Fire Impact to Water Quality and Stream Flow near Los Alamos National Laboratory: Results of Four Years of Monitoring. (Report prepared by Los Alamos National Laboratory, USA).

García-Lorenzo, M.L., Martínez-Sánchez, M.J., Pérez-Sirvent, C., 2014. Application of a plant bioassay for the evaluation of eco-toxicological risks of heavy metals in sediments affected by mining activities. J. Soils Sediments 14 (10), 1753–1765. http://dx.doi.org/10.1007/s1136.

Goforth, B.R., Graham, R.C., Hubbert, K.R., Zanner, C.W., Minnich, R.A., 2005. Spatial distribution and properties of ash and thermally altered soils after high-severity forest fire, southern California. Int. J. Wildland Fire 14 (4), 343–354. http://dx.doi.org/10.1071/WF05038.

González-Pérez, J.A., González-Vila, F.J., Almendros, G., Knicker, H., 2004. The effect of fire on soil organic matter—a review. Environ. Int. 30 (6), 855–870. http://dx.doi.org/10.1016/j.envint.2004.02.003.

Granged, A.J., Jordán, A., Zavala, L.M., Muñoz-Rojas, M., Mataix-Solera, J., 2011. Short-term effects of experimental fire for a soil under eucalyptus forest (SE Australia). Geoderma 167, 125–134. http://dx.doi.org/10.1016/j.geoderma.2011.09.011.

Harrison, J., Heijnis, H., Caprarelli, G., 2003. Historical pollution variability from abandoned mine sites, Greater Blue Mountains World Heritage Area, New South Wales, Australia. Environ. Geol. 43 (6), 680–687. http://dx.doi.org/10.1007/s00254-002-0687-8.

Hatten, J.Z.D., Scherer, G., Dolan, E., 2005. A comparison of soil properties after contemporary wildfire and fire suppression. For. Ecol. Manag. 220, 227–241. http://dx.doi.org/10.1016/j.foreco.2005.08.014.

Henig-Sever, N., Poliakov, D., Broza, M., 2001. A novel method for estimation of wild fire intensity based on ash pH and soil microarthropod community. Pedobiologia 45 (2), 98–106. http://dx.doi.org/10.1078/0031-4056-00072.

Hennessy, K., Lucas, C., Nicholls, N., Bathols, J., Suppiah, R., Ricketts, J., 2005. Climate Change Impacts on Fire-Weather in South-East Australia. Climate Impacts Group, CSIRO Atmospheric Research and the Australian Government Bureau of Meteorology, Aspendale.

Hernández, T., Garcia, C., Reinhardt, I., 1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol. Fertil. Soils 25 (2), 109–116. http://dx.doi.org/10.1007/s003740050289.

Hernandez, L., Probst, A., Probst, J.L., Ulrich, E., 2003. Heavy metal distribution in some French forest soils: evidence for atmospheric contamination. Sci. Total Environ. 312 (1), 195–219. http://dx.doi.org/10.1016/S0048-9697(03)00223-7.

Ignatavièius, G., Sakalauskienë, G., Oškinis, V., 2006. Influence of land fires on increase of heavy metal concentrations in river waters of Lithuania. J. Environ. Eng. Landsc. Manag. 14 (1), 46–51. http://dx.doi.org/10.1080/16486897.2006.9636878.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Inter-government Panel on Climate Change. http://www.ipcc.ch/report/ar5/wg1/, Accessed date: 10 October 2017.

Johnson, D.W., 1992. Effects of Forest Management on Soil Carbon Storage Natural Sinks of CO_2. Springer, pp. 83–120.

Jovanovic, V.S., Ilic, M., Markovic, M., Mitic, V., Mandic, S.N., Stojanovic, G., 2011. Wild fire impact on copper, zinc, lead and cadmium distribution in soil and relation with abundance in selected plants of Lamiaceae family from Vidlic Mountain (Serbia). Chemosphere 84 (11), 1584–1591. http://dx.doi.org/10.1016/j.chemosphere.2011.05.048.

Kabata-Pendias, A., 2011. Trace Elements in Soils and Plants. CRC Press, Taylor and Francis Group.

Keeley, J., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. Int. J. Wildland Fire 18. http://dx.doi.org/10.1016/j.earscirev.2011.08.002.

Kim, E.-J., Oh, J.-E., Chang, Y.-S., 2003. Effects of forest fire on the level and distribution of PCDD/Fs and PAHs in soil. Sci. Total Environ. 311 (1), 177–189. http://dx.doi.org/10.1016/S0048-9697(03)00095-0.

Kloprogge, J.T., Boström, T.E., Weier, M.L., 2004. In situ observation of the thermal decomposition of weddelite by heating stage environmental scanning electron microscopy. Am. Mineral. 89 (1), 245–248. http://dx.doi.org/10.2138/am-2004-0129.

Knoepp, J.D., Vsoe, J.M., Swank, W.T., 2008. Nitrogen deposition and cycling across an elevation and vegetation gradient in southern Appalachian forest. Int. J. Environ. Stud. 65, 389–408. http://dx.doi.org/10.1080/00207230701862348.

Krishna, A.K., Mohan, K.R., Murthy, N.N., Periasamy, V., Bipinkumar, G., Manohar, K., Rao, S.S., 2013. Assessment of heavy metal contamination in soils around chromite mining areas, Nuggihalli, Karnataka, India. Environ. Earth Sci. 70 (2), 699–708. http://dx.doi.org/10.1007/s12665-012-2153-6.

Kristensen, L.J., Taylor, M.P., Odigie, K.O., Hibdon, S.A., Flegal, A.R., 2014. Lead isotopic compositions of ash sourced from Australian bushfires. Environ. Pollut. 190, 159–165. http://dx.doi.org/10.1016/j.envpol.2014.03.025.

Lee, C.G., Chon, H.-T., Jung, M.C., 2001. Heavy metal contamination in the vicinity of the Daduk Au–Ag–Pb–Zn mine in Korea. Appl. Geochem. 16 (11), 1377–1386. http://dx.doi.org/10.1016/S0883-2927(01)00038-5.

Lee, J.-S., Chon, H.-T., Kim, K.-W., 2005. Human risk assessment of As, Cd, Cu and Zn in the abandoned metal mine site. Environ. Geochem. Health 27 (2), 185–191. http://dx.doi.org/10.1007/s10653-005-0131-6.

Liu, Y., Goodrick, S., Heilman, W., 2014. Wildfire emissions, carbon and climate: wildfire-climate interactions. For. Ecol. Manag. 317, 80–96. http://dx.doi.org/10.1016/j.foreco.2013.02.020.

Lottermoser, B.G., 2007. Mine Wastes: Characterization, Treatment and Environmental Impacts, 2nd ed. Springer, Berlin Heidelberg New York.

Machado, A., Serpa, D., Ferreira, R., Rodríguez-Blanco, M., Pinto, R., Nunes, M., Keizer, J., 2015. Cation export by overland flow in a recently burnt forest area in north-central Portugal. Sci. Total Environ. 524, 201–212. http://dx.doi.org/10.1016/j.catena.2004.09.006.

Maldon, 2017. Maldon History. Retrieved from. https://www.maldon.org.au/home/history (accessed 28 August 2017).

Mandal, A., Sengupta, D., 2006. An assessment of soil contamination due to heavy metals around a coal-fired thermal power plant in India. Environ. Geol. 51 (3), 409–420. http://dx.doi.org/10.1007/s00254-006-0336-8.

Marcos, E., Tárrega, R., Luis, E., 2007. Changes in a Humic Cambisol heated (100–500 °C) under laboratory conditions: the significance of heating time. Geoderma 138 (3), 237–243. http://dx.doi.org/10.1016/j.geoderma.2006.11.017.

Martin, R., Dowling, K., Pearce, D., Bennett, J., Stopic, A., 2013. Ongoing soil arsenic exposure of children living in an historical gold mining area in regional Victoria, Australia: identifying risk factors associated with uptake. J. Asian Earth Sci. 77, 256–261. http://dx.doi.org/10.1016/j.jseaes.2013.03.026.

Martin, R., Dowling, K., Pearce, D., Sillitoe, J., Florentine, S., 2014. Health effects associated with inhalation of airborne arsenic arising from mining operations. Geosciences 4 (3), 128–175. http://dx.doi.org/10.3390/geosciences4030128.

Martin, R., Dowling, K., Pearce, D.C., Florentine, S., Bennett, J.W., Stopic, A., 2016. Size-dependent characterisation of historical gold mine wastes to examine human pathways of exposure to arsenic and other potentially toxic elements. Environ. Geochem. Health 38 (5), 1097–1114. http://dx.doi.org/10.1007/s10653-015-9775-z.

Martin, R., Dowling, K., Nankervis, S., Pearce, D.C., Florentine, S., McKnight, S., 2017. In vitro assessment of arsenic mobility in historical mine waste dust using simulated lung fluid. Environ. Geochem. Health 1–13. http://dx.doi.org/10.1007/s10653-017-9974-x.

Mehes-Smith, M., Nkongolo, K., Narendrula, R., Cholewa, E., 2013. Mobility of heavy metals in plants and soil: a case study from a mining region in Canada. Am. J. Environ. Sci. 9 (6), 483–493.

Moody, J.A., Shakesby, R.A., Robichaud, P.R., Cannon, S.H., Martin, D.A., 2013. Current research issues related to post-wildfire runoff and erosion processes. Earth Sci. Rev. 122, 10–37. http://dx.doi.org/10.1016/j.earscirev.2013.03.004.

Moreira, F., Vaz, P., Catry, F., Silva, J.S., 2009. Regional variations in wildfire susceptibility of land-cover types in Portugal: implications for landscape management to minimize fire hazard. Int. J. Wildland Fire 18 (5), 563–574.

Moreno, B.G., Rousk, J., Bååth, E., 2015. Heated soil-water extract effect on bacterial growth: pH or toxic compounds? Flamma 6 (2), 42–45. 2015. http://hdl.handle.net/11441/50325.

Naidu, C., Srivasuki, K., 1994. Effects of forest fire on soil characteristics in different areas of Seshachalam hills. Ann. For. 2, 166–173.

Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland fire in ecosystems: effects of fire on soils and water. In: General Technical Report. RMRS-GTR-42-Vol. 4. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station (250 p).

Norouzi, M., Ramezanpour, H., 2013. Effect of fire on chemical forms of iron and manganese in forest soils of Iran. Environ. Forensic 14 (2), 169–177. http://dx.doi.org/10.1080/15275922.2013.781077.

Nunes, B., Silva, V., Campos, I., Pereira, J.L., Pereira, P., Keizer, J.J., Concalves, F., Abrantes, N., 2017. Off-site impacts of wildfires on aquatic systems—biomarker responses of the mosquitofish *Gambusia holbrooki*. Sci. Total Environ. 571–582, 305–313. http://dx.doi.org/10.1016/j.scitotenv.2016.12.129.

Odigie, K.O., Flegal, A.R., 2011. Pyrogenic remobilization of historic industrial lead depositions. Environ. Sci. Technol. 45 (15), 6290–6295. http://dx.doi.org/10.1021/es200944w.

Odigie, K.O., Flegal, A.R., 2014. Trace metal inventories and lead isotopic composition chronicle a forest fire's remobilization of industrial contaminants deposited in the Angeles National Forest. PLoS One 9 (9), e107835. http://dx.doi.org/10.1371/journal.pone.0107835.

Odigie, K.O., Khanis, E., Hibdon, S.A., Jana, P., Araneda, A., Urrutia, R., Flegal, A.R., 2016. Remobilization of trace elements by forest fire in Patagonia, Chile. Reg. Environ. Chang. 16 (4), 1089–1096. http://dx.doi.org/10.1007/s10113-015-0825-y.

Pardini, G., Gispert, M., Dunjó, G., 2004. Relative influence of wildfire on soil properties and erosion processes in different Mediterranean environments in NE Spain. Sci. Total Environ. 328 (1), 237–246. http://dx.doi.org/10.1016/j.scitotenv.2004.01.026.

Parkweb, 2018. Maldon Historic Area, Visitor Guide. https://parkweb.vic.gov.au/_data/assets/pdf_file/0011/315767/Park-note-Maldon-Historic-Area.pdf, Accessed date: 25 April 2018.

Parra, J.G., Rivero, V.C., Lopez, T.I., 1996. Forms of Mn in soils affected by a forest fire. Sci. Total Environ. 181 (3), 231–236. http://dx.doi.org/10.1016/0048-9697(95)05022-1.

Pearce, D.C., Dowling, K., Gerson, A.R., Sim, M.R., Sutton, S.R., Newville, M., et al., 2010. Arsenic microdistribution and speciation in toenail clippings of children living in a historic gold mining area. Sci. Total Environ. 408 (12), 2590–2599. http://dx.doi.org/10.1016/j.scitotenv.2009.12.039.

Pearce, D.C., Dowling, K., Sim, M.R., 2012. Cancer incidence and soil arsenic exposure in a historical gold mining area in Victoria, Australia: a geospatial analysis. J. Expo. Sci. Environ. Epidemiol. 22 (3), 248–257. http://dx.doi.org/10.1038/jes.2012.15.

Pereira, P., Ubeda, X., 2010. Spatial distribution of heavy metals released from ashes after a wildfire. J. Environ. Eng. Landsc. Manag. 18 (1), 13–22. http://dx.doi.org/10.3846/jeelm.2010.02.

Pereira, P., Ubeda, X., Outeiro, L., Martin, D., 2008. Solutes release from leaf litter (*Quercus suber*, *Quercus robur*, *Pinus pinea*) exposed to different fire intensities in a laboratory experiment. EGU Gen. Assem. http://dx.doi.org/10.1016/j.envres.2010.09.002.

Pereira, P., Ubeda, X., Martin, D.A., Guerrero, C., Mataix-Solera, J., 2009. Temperature effects on the release of some micronutrients from organic matter from Mediterranean forests. A comparison between laboratory experiment and prescribed fire. In: Paper presented at the International Meetings of Fire Effects on Soil Properties (02nd Edition).

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Guerrero, C., 2011. Effects of a low severity prescribed fire on water-soluble elements in ash from a cork oak (*Quercus suber*) forest located in the northeast of the Iberian Peninsula. Environ. Res. 111 (2), 237–247. http://dx.doi.org/10.1016/j.envres.2010.09.002.

Pereira, P., Ubeda, X., Martin, D., Mataix-Solera, J., Cerda, A., Burguet, M., 2013. Wildfire effects on extractable elements in ash from a *Pinus pinaster* forest in Portugal. Hydrol. Process. 28 (11), 3681–3690. http://dx.doi.org/10.1002/hyp.9907.

Pereira, P., Jordán, A., Cerdà, A., Martin, D., 2014. Editorial: the role of ash in fire-affected ecosystems. Catena 135, 337–339. http://dx.doi.org/10.1016/j.catena.2014.11.016.

Pino-del, J.N., Almenar, I.D., Rodríguez, A.R., Rodríguez, C.A., Rivero, F.N., Hernández, J.M., García, J.G., 2008. Analysis of the 1: 5 soil: water extract in burnt soils to evaluate fire severity. Catena 74 (3), 246–255. http://dx.doi.org/10.1016/j.catena.2008.03.001.

Plumlee, G.S., Martin, D.A., Hoefen, T., Kokaly, R., Hageman, P., Eckberg, A., et al., 2007. Preliminary analytical results for ash and burned soils from the October 2007 southern California wildfires. In: Open Report No. 2007 - 1407. US Geological Survey.

Prats, S.A., MacDonald, L.H., Monteiro, M., Ferreira, A.J., Coelho, C.O., Keizer, J.J., 2012. Effectiveness of forest residue mulching in reducing post-fire runoff and erosion in a pine and a eucalypt plantation in north-central Portugal. Geoderma 191, 115–124. http://dx.doi.org/10.1016/j.geoderma.2012.02.009.

Prats, S.A., dos Santos Martins, M.A., Malvar, M.C., Ben-Hur, M., Keizer, J.J., 2014. Polyacrylamide application versus forest residue mulching for reducing post-fire runoff and soil erosion. Sci. Total Environ. 468, 464–474. http://dx.doi.org/10.1016/j.scitotenv.2013.08.066.

Pyne, S.J., 2016. Fire in the mind: changing understandings of fire in western civilization. Philos. Trans. Soc. B371, 20150166. http://dx.doi.org/10.1098/rstb.2015.0166.

Quintana, J., Cala, V., Moreno, A.M., Parra, J., 2007. Effect of heating on mineral components of the soil organic horizon from a Spanish juniper (*Juniperus thurifera* L.) woodland. J. Arid Environ. 71 (1), 45–56. http://dx.doi.org/10.1016/j.jaridenv.2007.03.002.

Rabenhorst, M.C., 1988. Determination of organic and carbonate carbon in calcareous soils using dry combustion. Soil Sci. Soc. Am. J. 52, 965–968. http://dx.doi.org/10.2136/sssaj1988.03615995005200040012x.

Rayment, G.E., Lyons, D.J., 2014. Soil Chemical Methods - Australia. CSIRO Publishing, Collingwood, Victoria, Australia.

Ritson, P., Bouse, R., Flegal, A., Luoma, S., 1999. Stable lead isotope analysis of historic and contemporary lead contamination of San Francisco Bay estuary. Mar. Chem. 64 (1–2). http://dx.doi.org/10.1016/S0304-4203(98)00085-1.

Santín, C., Doerr, S.H., 2016. Fire effects on soils: the human dimension. Philos. Trans. Soc. B 371 (1696), 20150171. http://dx.doi.org/10.1098/rstb.2015.0171.

Santoro, A., A, H., Linsinger, T.P.J., Perez, A., Ricci, M., 2017. Comparison of total and aqua regia extractability of heavy metals in sewage sludge: the case study of a certified reference material. Trends Anal. Chem. 89, 34–40. http://dx.doi.org/10.1016/j.trac.2017.01.010.

Schaider, L.A., Senn, D.B., Brabander, D.J., McCarthy, K.D., Shine, J.P., 2007. Characterization of zinc, lead, and cadmium in mine waste: implications for transport, exposure, and bioavailability. Environ. Sci. Technol. 41 (11), 4164–4171. http://dx.doi.org/10.1021/es0626943.

Schaider, L.A., Senn, D.B., Estes, E.R., Brabander, D.J., Shine, J.P., 2014. Sources and fates of heavy metals in a mining-impacted stream: temporal variability and the role of iron oxides. Sci. Total Environ. 490, 456–466. http://dx.doi.org/10.1016/j.scitotenv.2014.04.126.

Sever, N., Poliakov, D., Broza, M., 2001. A novel method for the estimation of wildfire intensity based on ash pH and soil microarthropod community. Pedobiologia 45, 98–106. http://dx.doi.org/10.1078/0031-4056-00072.

Shakesby, R., 2011. Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth Sci. Rev. 105 (3), 71–100. http://dx.doi.org/10.1016/j.earscirev.2011.01.001.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth Sci. Rev. 74, 269–307. http://dx.doi.org/10.1016/j.earscirev.2005.10.006.

Shcherbov, B., 2012. The role of forest floor in migration of metals and artificial nuclides during forest fires in Siberia. Contemp. Probl. Ecol. 5 (2), 191–199. http://dx.doi.org/10.1134/S1995425512020114.

Sipos, P., Németh, T., Mohai, I., 2005. Distribution and possible immobilization of lead in a forest soil (Luvisol) profile. Environ. Geochem. Health 27 (1), 1–10. http://dx.doi.org/10.1007/s10653-004-1581-y.

Smith, H.G., Sheridan, G.J., Lane, P.N., Nyman, P., Haydon, S., 2011. Wildfire effects on water quality in forest catchments: a review with implications for water supply. J. Hydrol. 396 (1), 170–192. http://dx.doi.org/10.1016/j.hydrol.2010.10.043.

Someshwar, A.V., 1996. Wood and combination wood-fired boiler ash characterization. J. Environ. Qual. 25 (5), 962–972. http://dx.doi.org/10.2134/jeq1996.00472425002500050006x.

Soto-Jimenez, M.F., Hibdon, S.A., Rankin, C.W., Aggarwal, J., Ruiz-Fernandez, A.C., Paez-Osuna, F., Flegal, A.R., 2006. Chronicling a century of lead pollution in Mexico: stable lead isotope comparison analyses of dated sediment cores. Environ. Sci. Technol. 40 (3), 764–770. http://dx.doi.org/10.1021/es048478g.

Steenary, B.M., Karlsson, L.G., Lindqvist, O., 1999. Evaluation of the leaching characteristics of the wood ash and the influence of ash agglomeration. Biomass Bioenergy 16, 119–136.

Stein, E.D., Brown, J.S., Hogue, T.S., Burke, M.P., Kinoshita, A., 2012. Stormwater contaminant loading following southern California wildfires. Environ. Toxicol. Chem. 31 (11), 2625–2638. http://dx.doi.org/10.1002/etc.1994.

Stoof, C.R., Wesseling, J.G., Ritsema, C.J., 2010. Effects of fire and ash on soil water retention. Geoderma 159, 276–286.

Sultan, K., 2006. Distribution of Arsenic and Heavy Metals in Soils and Surface Waters in Central Victoria (Ballarat, Creswick and Maldon). PhD thesis. School of Science and Engineering, Federation University Australia.

Sultan, K., 2007. Distribution of metals and arsenic in soils of Central Victoria (Creswick-Ballarat), Australia. Arch. Environ. Contam. Toxicol. 52 (3), 339–346. http://dx.doi.org/10.1007/s00244-006-0050-2.

Tahiri, A.A., Laziri, F., Yachaoui, Y., El Allaoui, A., Tahiri, A.H., 2017. Heavy metals

leached from the waste from the landfill in the city of Meknes, and their impact on groundwater. J. Mater. Environ. Sci. 8, 1004–1014.
Taylor, C.J., Alley, W.M., 2001. Ground-water-level Monitoring and the Importance of Long-term Water-level Data. vol. 1217 United States Geological Survey (USGS).
Taylor, M.P., Mackay, A.K., Hudson-Edwards, K.A., Holz, E., 2010. Soil Cd, Cu, Pb and Zn contaminants around Mount Isa city, Queensland, Australia: potential sources and risks to human health. Appl. Geochem. 25 (6), 841–855. http://dx.doi.org/10.1016/j.apgeochem.2010.03.003.
Thomaz, E.L., 2017. Realistic soil-heating gradient temperature linearly changes most of the soil chemical properties. Soil Sci. Plant Nutr. 63 (1), 84–91. http://dx.doi.org/10.1080/00380768.2016.1255538.
Úbeda, X., Lorca, M., Outeiro, L.R., Bernia, S., Castellnou, M., 2005. Effects of prescribed fire on soil quality in Mediterranean grassland (Prades Mountains, north-east Spain). Int. J. Wildland Fire 14 (4), 379–384. http://dx.doi.org/10.1071/WF05040.
Úbeda, X., Pereira, P., Outeiro, L., Martin, D., 2009. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (*Quercus suber*). Land Degrad. Dev. 20 (6), 589–608. http://dx.doi.org/10.1002/ldr.930.
Ulery, A.L., Graham, R., 1993. Forest fire effects on soil color and texture. Soil Sci. Soc. Am. J. 57 (1), 135–140. http://dx.doi.org/10.2136/sssaj1993.03615995005700010026x.
Ulery, A., Graham, R., Amrhein, C., 1993. Wood-ash composition and soil pH following intense burning. Soil Sci. 156 (5), 358–364.
Ulery, A., Graham, R., Chadwick, O., Wood, H., 1995. Decade-scale changes of soil carbon, nitrogen and exchangeable cations under chaparral and pine. Geoderma 65 (1–2), 121–134. http://dx.doi.org/10.1016/0016-7061(94)00034-8.
UNEP, 2001. Summary Report: Abandoned Mine, Problems, Issues and Policy Challenges for Decision Makers. United National Environmental Programme, Geneva, Switzerland.
Verma, S., Jayakumar, S., 2012. Impact of forest fire on physical, chemical and biological properties of soil: a review. Proc. Int. Acad. Ecol. Environ. Sci. 2 (3), 168.
Verma, C., Madan, S., Hussain, A., 2016. Heavy metal contamination of groundwater due to fly ash disposal of coal-fired thermal power plant, Parichha, Jhansi, India. Cogent Eng. 3 (1), 1179243. http://dx.doi.org/10.1080/23311916.2016.1179243.
Vink, J.P., van Zomeren, A., Dijkstra, J.J., Comans, R.N., 2017. When soils become sediments: large-scale storage of soils in sandpits and lakes and the impact of reduction kinetics on heavy metals and arsenic release to groundwater. Environ. Pollut. 227, 146–156. http://dx.doi.org/10.1016/j.envpol.2017.04.016.
Violante, A., Cozzolino, V., Perelomov, L., Caporale, A., Pigna, M., 2010. Mobility and bioavailability of heavy metals and metalloids in soil environments. J. Soil Sci. Plant Nutr. 10 (3), 268–292. http://dx.doi.org/10.4067/S0718-95162010000100005.
Wade, D.D., Lunsford, J.D., Dixon, M.J., Mobley, H.E., 1989. A Guide for Prescribed Fire in Southern Forests. Technical Publication R8-TP-US Department of Agriculture. Forest Service, Southern Region (USA).
Wei, B., Yang, L., 2010. A review of heavy metal contaminations in urban soils, urban road dusts and agricultural soils from China. Microchem. J. 94 (2), 99–107. http://dx.doi.org/10.1016/j.microc.2009.09.014.
Westerling, A.L., Hidalgo, H.G., Cayan, D.R., Swetnam, T.W., 2006. Warming and earlier spring increase western US forest wildfire activity. Science 313 (5789), 940–943. http://dx.doi.org/10.1126/science.1128834.
Wilde, A., Bierlein, F., Pawlitschek, M., 2004. Lithogeochemistry of orogenic gold deposits in Victoria, SE Australia: a preliminary assessment for undercover exploration. J. Geochem. Explor. 84 (1), 35–50.
Wu, L., Taylor, M.P., 2017. Remobilization of industrial lead deposition in ash during Australian wild fires. Sci. Total Environ. 599–600, 1233–1240. http://dx.doi.org/10.1016/j.sciototenv.2017.05.004.
Wuana, R.A., Okieimen, F.E., 2011. Heavy metals in contaminated soils: a review of sources, chemistry, risks and best available strategies for remediation. ISRN Ecol. 2011. http://dx.doi.org/10.5402/2011/402647.
Zabala, L.M., Celis García, R., Jordán López, A., 2014. How wildfires affect soil properties. A brief review. Cuadernos de Investigación Geográfica 40 (2), 311–331. http://dx.doi.org/10.18172/cig.2522.
Zhang, X., Yang, L., Li, Y., Li, H., Wang, W., Ye, B., 2012. Impacts of lead/zinc mining and smelting on the environment and human health in China. Environ. Monit. Assess. 184 (4), 2261–2273. http://dx.doi.org/10.1007/s10661-011-2115-6.
Zhou, X., Zhao, Z., Zhang, J., Xue, X., 2010. Characteristics of heavy metal pollution in the soil around lead-zinc mining area. In: Paper Presented at the 04th International Conference on Bioinformatics and Biomedical Engineering, pp. 18–20 (June, 2010, Chengdu, China).
Zornoza, R., Carmona, D.M., Acosta, J.A., Martínez-Martínez, S., Weiss, N., Faz, Á., 2012. The effect of former mining activities on contamination dynamics in sediments, surface water and vegetation in El Avenque Stream, SE Spain. Water Air Soil Pollut. 223 (2), 519–532. http://dx.doi.org/10.1007/s11270-011-0879-5.